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Integrated Waste Management

Volume I

Edited by Sunil Kumar

INTEGRATED WASTE MANAGEMENT – VOLUME I

Edited by **Er. Sunil Kumar**

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Meet the editor

Er. Sunil Kumar is an expert in Environmental Engineering and Management with specialization in Solid and Hazardous Waste Management. He is a member of Editorial Boards of several international journals, such as Environmental Monitoring and Assessment, The Netherlands, and Open Waste Management Journal, UK. He is the Regional Editor for Asian countries in the International Journal of Process Wastes Treatment, Serial Publications, New Delhi, and is an expert member in Development of a Fact Base of Greenhouse Gas Emissions from Waste Sector in India leading up to 2030. He has served as Editor for several international publications, among which the Special Issue for Environmental Monitoring and Assessment, The Netherlands; Journal of Air and Waste Management Association, USA and International Journal of Environment and Pollution, and others.

Contents

Preface

The quantum of wastes generated in urban centres has become one of the difficult tasks for those responsible for their management. The problem is becoming acute specially in economically developing countries, where there is a financial crunch, and other resources are scarce.

Although, there are varieties of publications dealing with various topics of solid waste management, most of these documents have been published addressing the needs of developed nations. Only a few documents have been specifically written to provide the type of information that is required by those in developing countries. Also, most of the documents are not accessible to all the readers, and there as well a strong need to update the published documents once again in view of globalization. To maximize the use of limited available resources, it was decided to combine information gathered from both developed and developing countries on all the elements of solid waste management under the title "Integrated Waste Management". Due to overwhelming response from authors all around the world, the book has been divided into two parts, *i.e.* Volume I and II, and the papers have been grouped under different sub-headings.

This publication has been prepared primarily for researchers, engineers, scientists, decision-makers and policy makers involved in the management of solid wastes. The information provided in both the volumes would also be useful to students studying environmental science and engineering.

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Part 1

Planning and Social Perspectives Including Policy and Legal Issues

1

Governance Crisis or Attitudinal Challenges? Generation, Collection, Storage and Transportation of Solid Waste in Ghana

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1. Introduction

Waste is a continually growing problem at global, regional and local levels and one of the most intractable problems for local authorities in urban centers. With continuous economic development and an increase in living standards, the demand for goods and services is increasing quickly, resulting in a commensurate increase in per capita waste generation (Narayana, 2008). In most developing countries, the problem is compounded by rapid urbanization, the introduction of environmentally unfriendly materials, changing consumer consumption patterns, lack of political commitment, insufficient budgetary allocations and ill motivated (undedicated) workforce.

In Ghana, deficiencies in solid waste management (SWM) are most visible in and around urban areas such as Accra, Tema and Kumasi where equally important competing needs and financial constraints have placed an inordinate strain on the ability of the authorities to implement a proper SWM strategy in tandem with the rapid population growth. Consequently, most of the urban landscape is characterized by open spaces and roadsides littered with refuse; drainage channels and gutters choked with waste; open reservoirs that appear to be little more than toxic pools of liquid waste; and beaches strewn with plastic garbage. The insidious social and health impact of this neglect is greatest among the poor, particularly those living in the low-income settlements (UN-Habitat, 2010).

The provision of such environmental services had typically been viewed as the responsibility of the central government. However, the costs involved, coupled with the increasing rate of waste generation due to high urban population growth rates, have made it difficult for collection to keep pace with generation, thus posing serious environmental hazards. Apart from the unsightliness of waste, the public health implications have been daunting, accounting for about 4.9% of GDP (MLGRD, 2010a). Data from the Ghana Health Service indicate that six (6) out of the top ten (10) diseases in Ghana are related to poor environmental sanitation, with malaria, diarrhea and typhoid fever jointly constituting 70%-85% of out-patient cases at health facilities (MLGRD, 2010a).

Launching a National Campaign against Malaria in 2005, a Deputy Minister of Health noted that “malaria remains the number one killer in the country, accounting for 17,000 deaths, including 2,000 pregnant women and 15,000 children below the age of five”, a quarter of all child mortality cases and 36% of all hospital admissions over the past 10 years" (Daily Graphic, November 3, 2005: 11). The Ghana Medical Association also stipulates that about five million children die annually from illnesses caused by the environment in which they live (World Bank, 2007). In Kumasi, a DHMT Annual Report (2006) states that, "out of the cholera cases reported to health facilities, 50% came from Aboabo and its environs (Subin Sub-Metro) where solid waste management is perceived to be the worst".

Poor waste management practice also places a heavy burden on the economy of the country. In Accra, solid waste haulage alone costs the assembly GH¢ 450,000 (US$307,340) a month, with an extra GH¢ 240,000 (US$163,910) spent to maintain dump sites (Oteng-Ababio, 2010a), while in Kumasi, an average of GH¢720,000 (US$491,730) a month is spent on waste collection and disposal (KMA, 2010). The negative practice is also partly responsible for the perennial flooding and the associated severe consequences in most urban areas. The June 2010 flooding in Accra and Tema for example claimed 14 lives and destroyed properties worth millions of cedis (NADMO, 2011).

Admittedly, these tendencies are not exclusive to Ghanaian cities. Most urban centers in the developing world are united by such undesirable environmental characteristics. In Africa, it is anticipated that the worst (in terms of increasing waste generation and poor management practices) is yet to be experienced in view of the high rate of urbanization on the continent. By 2030, Africa is expected to have an urban population of over 50%, with an urban growth rate of 3.4% (UNFPA, 2009). The fear has been heightened by the changing dynamics of waste composition due partly to globalization and the peoples' changing consumption pattern. The increasing presence of non-biodegradable and hazardous waste types means that safe collection, transportation and disposal are absolutely crucial for public health sustainability.

The study examines how Accra, Tema and Kumasi, the most urbanized centers in Ghana, are grappling with SWM challenges in the wake of the glaring need to improve urban waste collection systems. It contributes to the menu from which practitioners can identify appropriate, cost effective and sustainable strategies for efficient solid waste collection, handling and disposal systems. Ultimately, the lessons learned from these experiences are useful not only for future policy formulation and implementation but more importantly, for other cities that are experimenting with private sector participation. Fobil et al (2008) intimated that, "the key observable feature is that the collection, transportation, and disposal of solid waste have moved from the control of local government authorities to the increased involvement of the private sector." It would be an understatement to say that understanding both the successes and failures of a city that has shifted most of the responsibility for SWM to the private sector is important for those planning to chart a similar course.

2. Study methodology

A variety of research methods were employed to achieve the objectives set. These included primary data collected using structured questionnaires, which covered the consumers, private providers of solid waste services, and local authorities in the three selected cities. The study also included a detailed investigation and survey of several collection points within each city. A detailed survey and investigation were performed to assess the current situation of the solid waste collection system in each of the cities. Also, selected focus group discussions were conducted with the executives of service providers, landlord associations as well as the rank and file of service beneficiaries, especially in the low-income areas. Other

secondary data sources were contacted, including some from the metropolitan assemblies, private organizations, and other community-based organizations.
To analyze the waste composition within each city, the entire area was examined based on their socio-economic characteristics (low, middle and high-income). A total of 25 houses in each city were randomly selected based on the population in each segment. Each selected house was provided with a 240-liter plastic waste bin, lined with a plastic bag. Residents were then required to dump their waste into the bin. Refuse from each house was collected twice a week, on Tuesdays and Thursdays, for eight weeks. The bags from each house were given special identification numbers and then transported to a designated site for segregation. A large clean plastic sheet was spread on the floor at the sorting site, and the contents were manually separated and the waste stream analyzed. Each category of waste for each house was weighed on a manual spring scale and recorded on a spreadsheet. The component materials in the waste stream were classified as follows:

- Organic (putrescible);
- Plastics (rubber);
- Textiles;
- Paper (cardboard);
- Metals and cans Glass.

The data was analyzed using a variety of tools and methods. Data collected from the interviews, investigations, surveys, and field work were processed, reviewed, and edited. The quantitative data were tabulated and relevant statistical tools and computer software were employed for analyzing and interpreting the results. Personal judgments, expert comments, and the results from the interviews and public survey were used as a basis for the analysis and interpretation of the qualitative data. In general, the results from the three locations— Accra, Tema and Kumasi— were virtually identical, therefore the analyses and subsequent discussions were organized and restricted around the main themes for the study area as a whole, with occasional references to a few exceptions for purposes of emphasis.

3. Result and data analysis

3.1 Waste generation

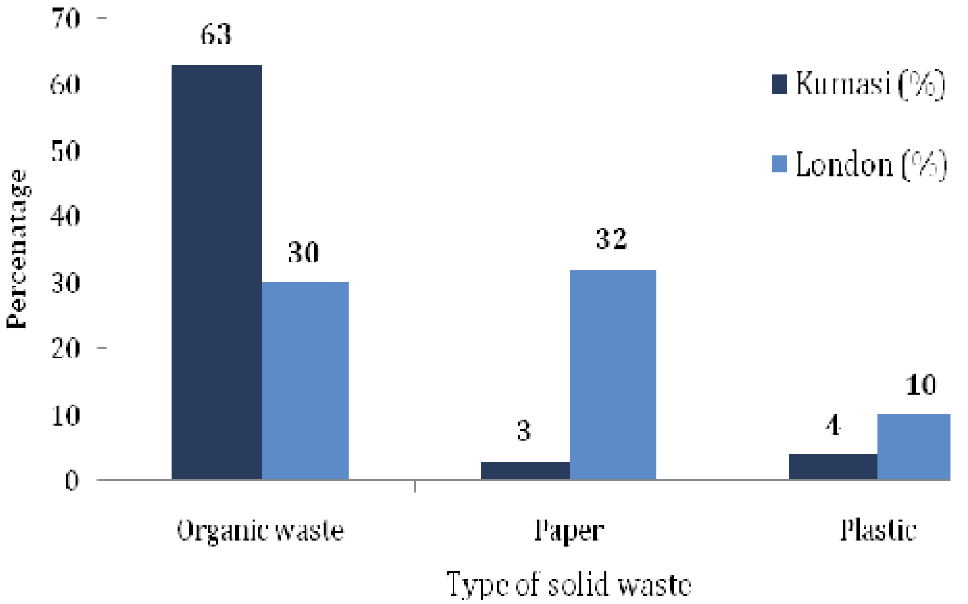
For the purpose of establishing the optimum collection systems, it is imperative to know the quantities and densities of the waste and where it is coming from. Generally, it is established that population growth greatly contributes to an increase in waste production. It has also been empirically established that waste generation has increased rapidly over the years. In Accra, for example, the amount of solid waste generated per day was 750-800 tonnes in 1994 (Asomani-Boateng, 2007); 1,800 tons per day in 2004 (Anomanyo, 2004); 2000 tons per day in 2007 (AMA, 2010; Oteng-Ababio, 2010a) and in 2010, it is estimated to be 2,200 tons (personal interview).
A dilemma relates to the amount of waste generated per person, which varies greatly with income (Houber, 2010). According to Blight and Mbande (1998), an affluent community may generate about 3 to 5 times as much waste per capita as a poor one. Boadi and Knitunen (2003) estimate that residents in low, middle and high income areas generate 0.40, 0.68 and 0.62 kilograms per day, respectively. They however noted that the density of waste is higher in low-income areas (0.50 per kilo liter) because their waste typically has a greater portion of organic and inert (sand and dust) matter, while packed products and cans form a significant

part of waste in high-income areas. Density of waste in high-income areas is estimated at 0.2 kilo per kilo liter while middle-income areas have 0.24 kilos per liter.

To date, there has not been any comprehensive empirical study on per capita waste generation in Ghana as a whole. All figures currently in use are crude estimates given by various authorities. Whilst the MLGRD, for example, gives the average daily waste generation as 0.51kg per person, the Water Research Institute (WRI) puts it at 0.41kg (WRI, 2000). Be that as it may, both have ramifications for planning purposes. Using these figures and the official as well as unofficial population of Accra for 2000, (i.e. 1.65 and 3 million, respectively), for example, in calculating daily waste generation, different figures are generated (i.e., between 841.5 and 1,530 tonnes based on MLGRD figures, and between 676.5 and 1,230 tonnes using the WRI figures). The disparities between these figures in a single year are just too great for any meaningful comparisons, analysis and proper planning, as a good statistical data is the link between good planning and good results. Despite the discrepancy, the low-income areas, home to about 80% of the population undoubtedly generate the bulk of solid waste in the study area.

3.2 Waste composition

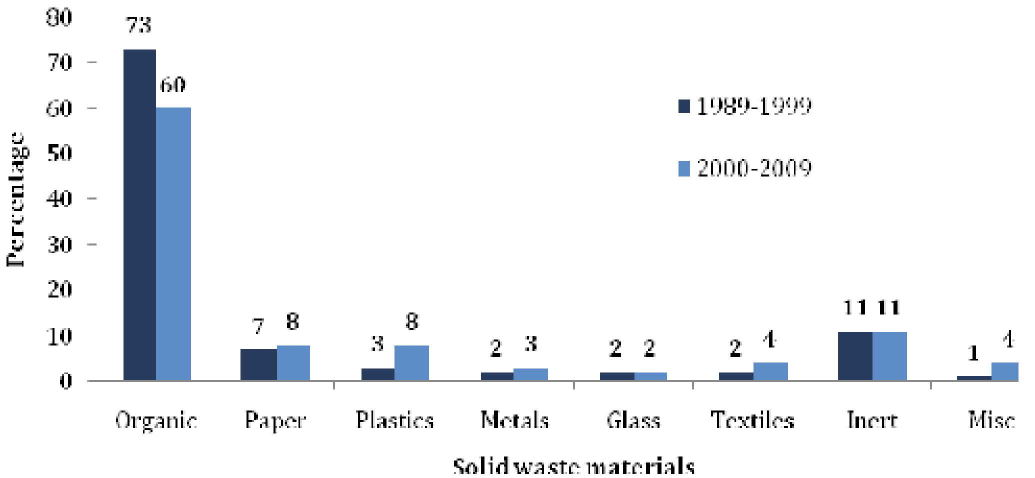
One significant aspect of solid waste in the study area is the changing complexity in the waste stream. Compared to the developed countries, wastes generated in the study area (and in developing countries for that matter) contain large volumes of organic matter. Table 1 presents a comparative study by Asase et al (2009) on the waste stream in Kumasi, Ghana and that in London in Ontario, Canada. The data show the clear difference between the composition of waste in the two cities, with organic materials accounting for 63% of waste in Kumasi but only 30% in London.



Source: Asase *et al*, 2009

Fig. 1. Comparison of waste streams in Kumasi (Ghana) and London (Canada)

According to Blight and Mbande (1998), the rapidly changing composition of waste stream in developing countries is a reflection of the dynamics of their culture, the per capita income of the community and the developmental changes in consumption patterns (Doan, 1998). Most residents have begun to make extensive use of both polythene bags and other plastic packaging, which creates an entirely new category of waste. Commenting on the menace of plastic waste in 2005, an Accra Mayor described it as "a social menace of a dinosaur, constituting over 60% of the 1,800 tons of waste generated within the Metropolis daily" (Daily Graphic, 2005: 28). Figure 2 compares the waste composition in Accra and Tema from 1989-1999 and from 2000-2009. Figure 2 shows a reduction of organic waste content from 73% in 1989-1999 to 60% in 2000-2009 while plastic surged from 3% to 8% within the same period. Also significant is the increasing miscellaneous category (which contains e-waste) from 1% in the 1990's to 4% in the 2010's. The emergence of e-waste in the waste stream is seen as an emerging challenge to waste management in Ghana (Oteng-Ababio, 2010b).

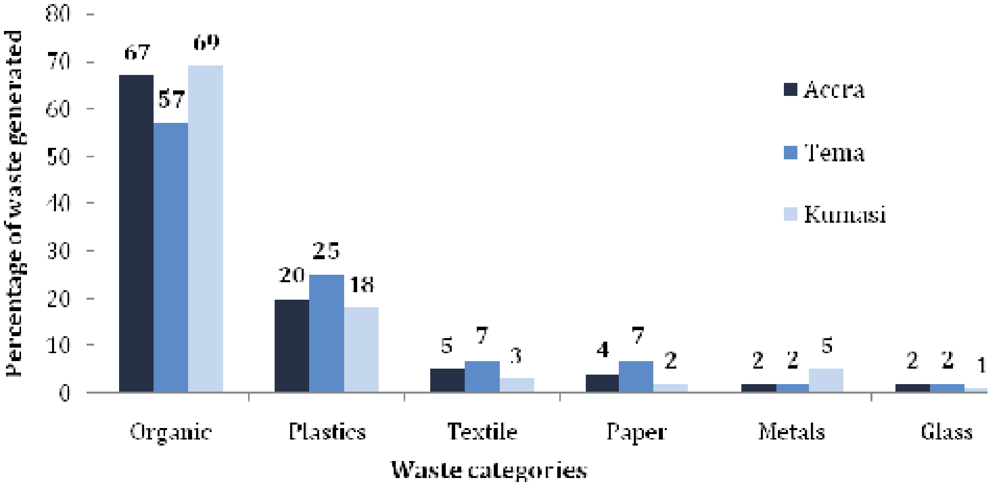


Source: Varying Composition of Solid Waste Stream, Greater Accra Metropolitan Area, Accra Metropolitan Assembly (AMA) (2004) and Baseline Survey, MMDAs (2008) in the National Environmental Sanitation Strategy and Action Plan (NESSAP), Ministry of Local Government and Rural Development (MLGRD), 2010. Note that there is a discrepancy in the above figures. The data from the period 1989-1999 adds up to 101% and not 100%. This data was taken directly from the source without changing this figure.

Fig. 2. Dynamics of Waste Composition-Accra/Tema (1989-1999 and 2000-2009).

Results of waste composition analysis conducted during the study were also consistent with the literature, with organic material (such as food, yard trimmings) being the most prevalent, comprising about 67% of the waste generated in all the three research localities (see Figure 3). Plastic material (such as plastic bottles and sachet bags) accounted for about 20%, while textiles accounted for about 5%. Figure 3 presents the percentage fractions of each category of the waste stream in the study areas.

From figure 3, it is clear that organic waste dominates the sampled waste stream while paper and plastic are the two other important constituents. The rest include glass, rubber, leather, inert materials (dirt, bricks, stones, etc.), wood, cloth, and other materials. It is estimated that the percentage contribution of most waste constituents will remain close to those of present years; however, there will be a dramatic change in plastic waste production due to the increased use of plastic products among the Ghanaian populace, especially people in the major cities. From the foregoing urban waste classifications, it is evident that different categories of waste may require different handling, collection and disposal strategies. The successful implementation of any SWM system is partly dependent on the synergy between waste storage, loading and transportation. The compatibility between these three elements of SWM systems ensures efficient and sustainable operations. Generally, an appropriate waste storage facility must satisfy many requirements, including convenience, size and durability.



Source: Field Survey, 2010.

Fig. 3. Percentage fractions of each category of the waste stream in the study areas

3.3 Waste storage practices

The research identified two major modes of storage for household solid waste in the study area. The first involves the use of polythene bags, card board boxes, and old buckets, which was quite prevalent in both the low and middle-income areas, and the standard plastic containers in the high-income neighborhoods. It further revealed that the more improvised (unorthodox) systems are used for waste storage, the more likely the area suffers poor SWM practices. A critical analysis of the mode of solid waste storage in the various residential areas within the study area buttresses this. The situation in Accra is presented in Table 1.

Table 1 shows the various waste storage facilities used in residential areas in Accra. Seventy-four (82.2%) respondents in the high-income areas, where the house-to-house (HH) system is prevalent, use the standard plastic containers. Only 16 (17.78%) of them used unorthodox methods (polythene bags, old buckets, etc.). This is perhaps because waste in the high-income areas is collected once or twice a week and therefore needs to be properly stored. Alternatively, in the low-income areas where the communal container collection (CCC) operates, 155 (73.81%) respondents used impoverished storage facilities. Indeed, waste in such areas is stored for a very short period and residents can visit the container sites more than once a day. A chi-square test, conducted on the mode of waste storage and the residential location, gave a value of 105.579 at 6 degree of freedom (df). By inference, there is a highly significant relationship between the mode of waste storage and residential areas, and by extension, the wealth of the area, and this is consistent with the literature (UN-Habitat, 2010). This is also a function of the mode of waste disposal.

Classification	Std. Containers		Polythene Bags		Pile outside		Others		Total	
	Freq.	%	Freq.	%	Freq.	%	Freq.	%	Freq.	%
Low Class	44	20.95	155	73.81	10	4.76	1	0.48	210	100
Medium Class	91	45.5	93	46.5	13	6.5	3	1.5	200	100
High Class	74	82.22	16	17.78	0	0	0	0	90	100
Total	209	41.8	264	52	23	4.6	1	0.2	500	100

Source: Field Survey, 2010
Note: Chi-square value 105.579. Asyump. Sig. (2-Sided) 0.000 a. 4 cells (33.3%) were expected count less than 5. The minimum expected count is 0.72.

Table 1. Mode of Solid Waste Storage by Residential Areas in AMA

Similar observations were made in Tema and Kumasi. However, unlike Accra and Kumasi, 100% and 65% of respondents in the high and middle-income areas of Tema, respectively, used the standard plastic containers. This is primarily due to the planned nature of Tema. Additionally, the authorities in Tema, through the Waste Management Department (WMD), have been supplying plastic containers to residents at a fee, thus providing motivation and impetus for the use of the standard containers. Consequently, littering in these areas is relatively minimal and thus the city has a relatively clean environment.

It can also be inferred from the study that most residents in the low-income areas generally lack the economic capabilities and will typically not willingly spend much money on waste storage containers. This situation is more likely to occur under the container system where children can send waste to the container site at least twice in a day. Indeed, because children are mostly involved in waste disposal, residents are compelled to use lighter containers like polythene bags, instead of the costly but 'heavy' standard containers, which are somewhat incompatible with the prevailing institutional arrangement. Another observation, especially in Kumasi, was that although some middle-income residents claim to be using standard containers, practical observation revealed the use of improvised galvanized containers (see Figure 4), possibly due to the higher cost of the former. A market survey of the prices of the standard containers revealed that a 120-liter container (see Figure 4) which was GH ¢3.3 in 1995, cost GH ¢150 in 2010, an increase of about 4,445%, while a galvanized container was selling at only GH ¢ 15.

The use of unapproved storage facilities and children in waste disposal, especially in the low-income areas, presents its own problems, which the authorities seem to have glossed over. For example, in most cases, children find it difficult to properly access the containers because of their height. It thus becomes more convenient to throw waste on the ground instead of dumping it in the refuse container. The situation is even worse in areas where they are supposed to be assisted by caretakers for a fee, which is reminiscent of the "Pay-As-You-Dump" (PAYD) system. Additionally, these unorthodox containers are constantly subjected to ransacking by domestic animals to the detriment of the environment. This ultimately results in indiscriminate littering at the sites, with its attendant poor hygienic conditions (see Figure 5). It is therefore not uncommon to see scattered waste bags being loaded into the collection trucks with the help of shovels and rakes. This is a slow, laborious and unhygienic system that results in poor vehicle utilization and low labour productivity.

Source: Field Survey, 2010

Fig. 4. Samples of standard and improvised waste receptacles.

Source: Field Survey, 2010.

Fig. 5. Indiscriminate dumping at a container site. (Note the high presence of plastics)

It was further observed that in the middle-income areas of Accra and Kumasi, some residents in the informal sector, including mechanics, sawmill operators, car washing bays and chop bar operators are virtually compelled to use unofficial dumping practices because of the nature and volume of the waste they generate vis-à-vis the cost of disposing such waste. In other words, the socio-economic characteristics in those neighborhoods make the official services virtually inaccessible (cost prohibitive) to these residents, thus buttressing the argument that social propinquity may be different from social accessibility (Phillips, 1981; 1984).

3.4 Waste collection

For any sustainable waste management system, the collection system must be designed and operated in an integrated way. In particular, the method of loading a collection truck must suit the mode of storing waste. If the waste is destined for recycling, then it should be designed to ensure minimum contamination. It is also important to ensure that, if waste is to be deposited in a landfill, then the trucks in operation should be appropriate for landfill manoeuvring. Generally, the waste collection rate in most African cities has been typically low (see Table 2), ranging from 40-50% (Mwesigye et al, 2009). However, available data in the study area indicate that there has been a significant increase in total waste collection with the introduction of the private sector. Accra, for example, is currently said to have attained a collection rate of 70% (AMA, 2009; Oteng- Ababio, 2010a) or 80% (Huober, 2010). The remaining 20-30% uncollected waste is either burned or buried or dumped indiscriminately (MLGRD, 2010b).

	Population	Growth (%)	% of solid waste collected
Abidjan (Cote D'Ivore)	2,777,000	3.98	30-40
Dakar (Senegal)	1,708,000	3.93	30-40
Ndjamena (Chad)	800,000	5.00	15-20
Nairobi (Kenya)	2,312,000	4.14	30-45
Nouakchott (Mauritania)	611,883	3.75	20-30
Lome (Togo)	1,000,000	6.50	42.1
Yaoundé (Cameroon)	1,720,000	6.80	43
De res Salam (Tanzania)	2,500,000	4.30	48

Source: Sotamenou 2005 for Yaoundé; Rotich et al 2006 for Nairobi; Benrabia 2003 and Bernard 2002 for Dakar and Abidjan; EAMAU 2002 for Lomé; Doublier 2003 for Ndjamena; Ould Tourad et al 2003 and Pizzorno Environnement for Nouakchott; and Kassim 2006 and the International Development Research Centre for Dar es Salaam in Parrot et al 2009 (cited in Houber, 2010).

Table 2. Cross-country analysis of population, growth and solid waste collected

Table 3 presents the trend of total volume of waste collected between 2002 and 2008 in Accra. The data indicate an overall progressive improvement in the collection rate from 476,281.92 in 2002 to 658,044.06 in 2008, an increase of about 38%. What remains debatable is whether such increase has translated into quality service delivery. One would have expected much improved sanitation, especially in low-income areas, yet ironically, the opposite pertains. Those areas continue to be engulfed in filth and this seems to give credence to the perception that some WMD officials collaborate with some private service providers to cheat the assemblies. The data however shows a steady decline in the volume of waste collected between 2002 and 2004. The situation could be attributed to the tardy payments from the authorities, which was worsened by the sharp increase in fuel prices and other operational costs at the time.

Year	Waste Generated	Waste collected	Waste uncontrolled	% collected	Private contractors shares
2002	675,000.00	476,281.92	198,718.08	70.56	N/A*
2003	657,000.00	419,671.30	237,328.70	63.88	N/A*
2004	657,000.00	424,802.42	232,197.58	64.66	96.28
2005	657,000.00	512,030.95	144,964.05	77.93	98.46
2006	657,000.00	639,854.69	17,145.31	97.39	98.06
2007	730,000.00	604,756.43	125,243.57	82.84	99.73
2008	730,000.00	658,044.06	71,955.94	90.14	99.94

Source: AMA/WMD, 2009. * N/A means data was not available at the time of the study.

Table 3. Waste Generation and Collection in Accra (2002-2008)

The study reveals that waste collection services within the study areas are provided by one of two means: the house-to house (HH) and the communal container collection (CCC) systems. The HH system is designed to serve low-density, medium and high-income areas that have easy access and identifiable houses. With this system, private waste collectors are expected to pick up waste from private homes, expectedly with compact trucks, for dumping. The CCC, on the other hand, is designed to serve high-density, low and middle-income areas that are more difficult to access by road. Under this system, residents are expected to carry their waste free of charge to a communal container that is later emptied by a collection truck. The assembly is expected to pay the private waste collector GH ¢10 per every ton of waste sent to the dump site. Table 4 presents a brief discussion on the characteristics of the two institutional arrangements.

The study revealed that because the low-income areas offer fewer opportunities for profit (due partly to the tardy payment of the assemblies) compared to the high-income areas where service providers have the privilege of negotiating directly with service beneficiaries, the former generally receive the lowest priority from the service providers. There is also enough evidence to suggest that the communal containers provided by the authorities are frequently inadequate in terms of their volumes and the population threshold they are expected to serve. This happens in situations where un-emptied or overflowing communal containers have become common sights in such areas, constituting both a nuisance and health hazards. The situation has worsened due to the high organic and moisture content of the waste as well as the generally high temperatures which facilitate rapid decomposition, coupled with the fact that waste in those areas is often mixed with human waste due to inadequate sanitation facilities. The problem of inadequate facilities does not only lead to indiscriminate dumping of waste, but also to strong foul smells emanating from the waste, both of which compromise the health needs of residents.

The limited refuse containers compel residents to travel long distances to access the few in circulation. During the study, only 10% and 8% of respondents from the low-income areas in Accra and Tema, respectively, indicated they traveled within a 50 meter radius to waste container sites to dispose of their waste. The rest have to travel beyond the 50 meter mark up to over 200 meters to assess a container. The situation appears worse in Kumasi where about 50% of residents in Aboabo (low-income area) had to travel over 150 meters to the nearest refuse receptacle. This has a negative impact on solid waste disposal as most residents have the tendency of finding other convenient places to dispose of their waste, places which are normally very close to where they live.

Variables	**House-to-house collection**	**Collective container collection**
Standard collection frequency	At least Weekly	Daily
Dominant waste storage container	Standard Plastic bins	Old buckets Polythene bags
Mode of transporting solid waste	Multi-lift truck Open truck Three-wheeled tractor Pushcart	Skip-loader
Mode of lifting waste bins/containers	Multi-lift trucks	Skip-loader
Main areas of operation	High income Middle income	Low income Middle income
Characteristics of areas	Good road-network Excellent accessibility to houses	Poor road network Poor accessibility to houses
User fees	Yes	No
Service provided by	Private sector	Local authority/Private sector
Private contractor pay dumping fees to AMA	Yes	No

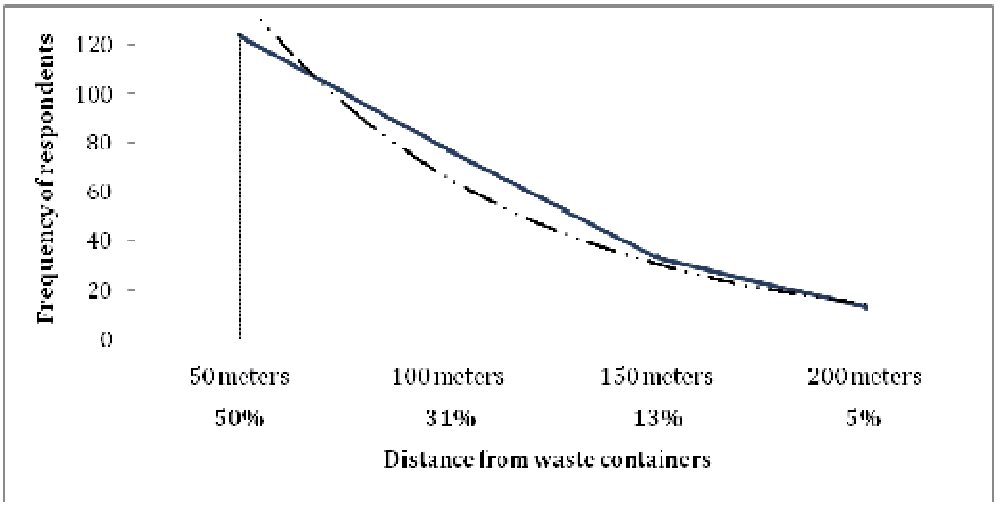
Source: Field Survey, 2010

Table 4. Major Characteristics of the Institutional Arrangements in the study area

Many reasons might have accounted for this development. For example, it was established that some residents have encroached on the container sites, while the inability of the authorities to regularly pay the collection companies adversely affects the rate at which the containers are emptied. Consequently, some residents who cannot stand the filth and the attendant stench vehemently protest and resist the continuous location of the containers at those sites. During the survey period, there were instances where some residents of Abossey Okai, for example, physically attacked the workers of Golden Falcon Company (a private waste collection company) for attempting to place a container at a particular spot.

3.4.1 Distance-decay and the use of refuse containers

Attempts were also made to ascertain how far residents are prepared to travel to access a refuse container. In Accra, 124 (50%) respondents in low-incomes areas indicated their willingness to access communal containers within a 50 meter radius while only 13 (5%) are prepared to travel about 200 meters for the same purpose (see Figure 6). The situation was not different from the responses from Kumasi, though fewer people (only 3.7%) were prepared to travel beyond the 50 meter radius. This is due to the drudgery and opportunity cost associated with commuting long distances daily to the container site. By inference, the longer the distance, the more people are likely to abuse the system, thereby legitimizing the principles enshrined in the distance-decay theory. The long distances and the fact that in most instances the containers will be over-flowing on arrival, serve as deterrents to residents who then use any available open space as an alternative dumping site. From the foregoing, it can be deduced that, there is a maximum travel threshold within which residents will voluntarily access the communal container. Once this is exceeded, utilization tends to fall considerably. This negative relationship observed is reinforced empirically by the very little littering in areas serviced by HH operators, where wastes are virtually collected at the doorsteps of residents, as against the container system where residents have to travel long distances and unsightly scenes have become the bane of the society, as is the case in Nima in Accra, Ashaiman in Tema and Aboabo in Kumasi.



Source: Field Survey, 2010

Fig. 6. Distance-Decay in residents' willingness to access the nearest refuse container

3.4.2 The role of the informal sector in waste collection

The study revealed the use of the services of Kaya Bola[1] in the waste collection system, especially in Accra and Tema. In Accra, such activities are confined to the middle and high-income areas while in Tema, it is predominantly in the low-income areas. The fact is that in Tema, the middle and high-income areas are well planned and therefore facilitate the HH operation, which is generally seen as quite efficient and acceptable. On the other hand, the

[1] Kaya Bolas or Truck Boys are porters who carry solid waste from residences, markets and offices in sacks, baskets, on trucks, etc to a container or dumping sites for a fee.

middle-income areas in Accra and Kumasi did not have that advantage. Some residents are thus compelled to complement their official "unsatisfactory" services with those of Kaya bola.

The middle-income areas of Abossey Okai, Adabraka and Kaneshie in Accra, for example, are officially supposed to be serviced under the container system. However, the study revealed about 25% of respondents in each of these neighborhoods use the services of Kaya Bola. This has been necessitated by the fact that these areas have essentially become part of the commercial hub of Accra and presumably, contain some modestly rich residents. Consequently, because of their commercial interests and wealth, they can afford the services of Kaya Bola as a trade-off for the apparent inefficiency of the formal institutional arrangements. The service was quite noticeable in the low-income areas where, due to its peculiar infrastructural challenges, the official HH system is rendered technically impossible. In such circumstances, the few affluent people rely on the services of the Kaya bola to meet their environmental needs.

Be that as it may, the activities of the Kaya Bola cannot be a panacea to the solid waste menace confronting the city authorities. Indeed, their present modus operandi actually contribute to the creation of filth, especially around the container sites, the reason being that their activities are unofficial and therefore are not properly integrated into the overall SWM system.

They also do not have the mechanism to off-load their collected waste into the already over-flowing containers. In the process, they litter the sites or find other means to dispose of the collected refuse which, in most cases, is inimical to environmental and societal health. The city authorities should therefore make attempts to incorporate these activities into other CBO operations or harness them to formally provide HH services to the official operators of the CCC. The idea should not be to roll them into the tax bracket but to structure their activities and let them provide checks on their colleagues.

3.5 Technology for solid waste collection

All things being equal, the mode of waste storage and disposal influences the technology used for collection (Obiri-Opareh, 2003). Generally, the result of this study confirms this observation, though it also reveals that some service providers in all the cities use unorthodox technology to execute their contracts. Among the reasons assigned for this development is the authorities' inability to adequately resource them (private contractors) or make prompt payments for services provided. Under the current HH system, contractors use multi-lift trucks, open trucks, three-wheeled tractors and power tillers. In the CCC system, however, skip-loaders were very predominant because of the use of central containers.

By inference, there is a correlation between the type of technology used and the material wealth as well as the layout of an area. It was however observed that most of the available trucks often showed signs of heavy wear with a limited useful economic life. Even the number of these trucks in circulation vis-à-vis the job at hand was very limited. Most of the trucks had also broken down and were stripped of spare parts due to the difficulty and cost in buying new parts (see Figure 7).

Interactions with some private contractors, including Golden Falcon and ABC Waste, revealed that most of the supposed faulty trucks only needed a part to be operational. However, because of the irregular payment from the authorities and the fact that many of the parts are not locally accessible, many of the companies overlooked them and put in circulation the few for which they could provide fuel and other lubricants. The same reason also explains why some contractors are compelled to use rickety vehicles, especially in the less-privileged suburbs, as captured in Figure 8. In terms of equipment holdings, Accra is the worst off among all the cities. The assembly has no road-worthy vehicle for waste collection, apparently because it has fully privatized its solid waste collection services.

Source: Field Survey, 2010

Fig. 7. Some disused trucks for some private solid waste companies in Accra

(Note the plastic waste which has started accumulating around the truck).
Source: Field Survey, 2010

Fig. 8. A broken down refuse collection Bedford truck (GR5308B) in Accra.

3.6 Area of coverage of waste collection

One cardinal objective for introducing private sector participation in SWM was to help improve the aerial extent of efficient service provision. However, empirical data from the survey could not wholly support this. For instance, over 30% of residents in Kumasi still had no official institutional arrangements for waste collection and therefore continue to practice crude dumping. In Accra, the current total waste collection coverage is about 70%. The remaining 30% is collected either irregularly or not at all (Oteng-Ababio, 2010a). About 10% of Tema is still rural and services, where they existed, were poor. Even the appropriateness of these figures is in doubt, in view of the increasing number of households over the past years, a situation which has led to the rapid deterioration of waste management facilities that are not replaced and to the increasing amount of waste generated by street sweepers, industrial areas and the central business district (CBD).

The conventional municipal SWM approach based on collection and disposal has failed to provide the anticipated efficient and effective services to all residents. In Tema, the collection coverage is estimated to be 65% while the rest are dumped indiscriminately into drains and gutters (Post, 1999; Oteng-Ababio, 2007). Probably, the un-serviced neighbourhoods are not experiencing the kind of filthy environment that pertains in Nima (Accra), Ashaiman (Tema) and Aboabo (Kumasi) because the nature and volume of waste in the fringe communities are more biodegradable and can be handled by the eco-system. However, the recent increasing use of plastics is gradually posing serious health and environmental threats to the otherwise uninterrupted natural way of managing the fringe environment.

3.7 Waste transportation

The main objective of any waste collection system is to collect and transport waste from specific locations at regular intervals to a disposal site at a minimum cost. In this regard, many technical factors have a direct bearing on the selection of a collection system and vehicles for any particular situation. In other words, the choice of vehicle and storage system are closely related. Among the factors influencing the choice of a possible transportation (vehicle) include the rate of waste generation; density; volume per capita; constituents; transport distance and road conditions. Others include traffic conditions, the level of service, and beneficiaries' willingness to pay. The study revealed that among the commonest means of transportation used in the study area are handcarts, pushcarts and wheelbarrows. These are used to carry waste over short distances. In addition, carts drawn by bullocks, horses or donkeys have been used to pull relatively larger loads. These appear appropriate especially in the densely populated, inaccessible low-income areas with serious traffic congestion. Unfortunately, the study reveals that city authorities and most residents currently perceive this system as primitive and therefore abhor it.

The exclusive use of "sophisticated" vehicles, ranging from tractors to specifically designed trucks, normally at the behest of donor agencies or "corrupt" city authorities, have become the order of the day, notwithstanding the obvious institutional, financial and infrastructural challenges and the varying areal differentiation. For example, in 1997, AMA entered into a financial agreement with the Ministry of Finance for a line of credit for US$14,630,998 from Canada's EDC to purchase waste collection vehicles. Most of the said vehicles had been parked by 2000 due to lack of spare parts and maintenance know-how (Oteng-Ababio, 2007). Thus, technically, the low technology options such as donkey carts, pushcarts are deemed appropriate and convenient for deployment in densely populated, inaccessible neighbourhoods while the high technology ones like skip-loaders and compaction trucks can operate in more accessible areas.

Besides, it is relatively easy to acquire and maintain the low technology options, though they have the tendency of compromising environmental sustainability if they are not properly integrated into the overall SWM programme. By inference, it can be concluded that to ensure any sustainable efficient waste collection system the transportation mechanism and equipment must meet the varying needs of the urban space. It must also be affordable and easy to operate and maintain, with ready availability of spare parts on the local market. Sophisticated imported equipment, mostly procured through donor support, has often not lasted long, quickly becoming moribund and creating equipment graveyards at the local authority depots (see figure 9).

Source: Field Survey, 2010

Fig. 9. Some disused trucks of AMA at the Assembly's depot.

4. Some problems affecting SWM in GAMA

The study has identified a clear relation between the SWM practices and cleanliness. It also demonstrated that although a greater part of the study area is fairly clean, especially the high income and some middle-income areas, the low-income areas are filthy due to poor SWM practices, occasioned by the high population growth (Obiri-Opareh, 2003; Awortwi, 2001) and the changing nature and composition of waste (Doan, 1998). Additionally, most high-density, low-income areas where about 60% of the city's waste is generated are poorly accessible by road. This makes the removal of accumulated waste using motorized vehicles difficult, hence the use of the container system, which is also fraught with many problems. For example, the fact that some residents have to travel long distances to access waste receptacles encourages indiscriminate littering. Furthermore, inadequate funding and poor cost recovery capabilities have resulted in acute financial problems for the authorities. The situation has been aggravated in Accra and Kumasi where the container system, which caters for almost 60% of total waste collection, is fee free, thus putting severe financial constraints on the authorities which invariably affects service delivery.

The inability of the assemblies to enforce their own by-laws also impacts negatively on SWM. For example, a key statutory document required for the proper development of any city environment is a building permit. This is to ensure decent and safe buildings in an orderly manner across urban space. However, for many developers, obtaining this license has been a nightmare. Accordingly, most developers aware of the inconveniences deliberately flout the rules, at times with the connivance of some officials. Hence, the many unplanned, haphazard neighborhoods which hinder proper waste collection. An equally important observation is the authorities' inability to involve all stakeholders in the decision making process and build on consensus. Apart from autocratically deciding which institutional arrangement operates in which neighborhoods, decisions regarding the waste collection vehicles that are supplied are often made by the authorities who have very little understanding of technical issues and are therefore devoid of operation and performance competence. Donor agencies are sometimes also guilty of providing vehicles with inappropriate design and from a manufacturer almost unknown to the region where the vehicle is expected to be maintained. In such situations, sustainability of the vehicles and service delivery is compromised.

Certain lax attitudes of some residents and officials have also contributed to poor SWM practices. For instance, although most residents yearn for refuse containers to dispose of their waste, they simultaneously object to the location of such containers near their houses, under the pretence that the sites are not properly maintained and/or the containers are not emptied on time, creating spillage and foul stench. The attitude of some officials also indirectly helps perpetuate the problem. For example, a key attitudinal factor that has engendered the growth of undesirable settlements like Sodom and Gomorrah, Ashaiman, and Aboabo is the quick provision of state-sponsored utility services and infrastructural facilities like water, electricity and telephone. Additionally, the massive encroachment on public lands constrains the authorites' ability to find an appropriate place to locate refuse containers. The Ghanaian media are replete with news about such abuses.

5. Waste collection dilemma in Ghana: a governance crisis or attitudinal challenge?

The study provides an overview of solid waste storage, collection and transportation in three Ghanaian cities. The result clearly shows that the present situation leaves much to be desired. Faced with rapid population growth and changing production and consumption patterns, the authorities, like those in many cities in Sub-Saharan Africa, are seriously challenged to implement the infrastructure necessary to keep pace with the ever increasing amount of waste and the changing waste types. Although the waste collection rate has improved over the past decade due to greater private sector participation, waste services in low-income areas are still inadequate. Admittedly, many factors jointly account for this: institutional weakness, inadequate financing, poor cost recovery, the lack of clearly-defined roles of stakeholders and the lax attitude of officials and residents.

However, a critical analysis of these challenges reveals a fundamental cause which is skewed towards a governance crisis rather than attitudinal challenges. For example, policies relating to the adaptation of institutional arrangements and the purchasing of transportation equipment are developed in the absence of both the private sector and public participations. Such unilateral decisions ignore the realities of local conditions, as in the case of the failure to acknowledge the operations of the Kaya bola. The authorities have also failed to implement the necessary by-laws to make compliance with policies enforceable. For example, citizens in poor neighborhoods may simply refuse to pay for waste services and begin to dump waste indiscriminately, creating financial challenges for service providers who will then be compelled to downgrade the quality of service. This will in turn possibly frustrate the fee paying residents in the middle and high-income areas. Such policy inconsistencies have created deep fissures in the relationship between the authorities and a large segment of the citizenry, culminating in loss of trust and confidence as well as some lax attitudes and behaviour as are being exhibited in some neighborhoods.

Revising this trend will be a daunting task. The authorities need to change from these current out-oriented, foreign-inspired policies. They need to look inward and adopt an all-inclusive, creative and experimental approach that takes into consideration local conditions and engages the public in a democratic manner. Moving towards a genuine participatory approach to waste management will not come on a silver platter. It calls for a paradigm shift on the part of the authorities and it will take time to win public interest, acceptability and participation. Probably as a first step, the authorities need to formalize and integrate the operations of the hitherto neglected informal sector into the overall SWM system. The sector does not only provide services for the almost neglected low-income neighbourhoods (home to about 70% of the urban population) but also serves as a source of livelihood for thousands of urban poor. Streamlining such operations will therefore create public confidence and also avert any environmental repercussions of their operations. At the end of the day, it is the poor management of waste, not the waste per se, that makes the cities filthy.

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2

Institutional Matrix for Sustainable Waste Management

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1. Introduction

1.1 Background information

Ensuring environmental sustainability is one of the eight Millennium Development Goals (MDGs) adopted by world leaders in 2000. This underscores the priority the world places on achieving and maintaining a clean and safe environment for both present and future generations. Waste management is therefore a subject of immense interest to all nations and peoples.

Significant differences, however, characterise waste management service delivery in developed and developing countries. Though the developed countries generate larger amounts of wastes, they have developed adequate facilities and strong institutions to manage them. Developing countries, on the other hand, are faced with an uphill task of providing adequate facilities for waste management and to ensure their sustainable operation and maintenance. The Millennium Development Goals Report for 2007 notes that half the population of the developing world lack basic sanitation and that in order to meet the MDG target, an additional 1.6 billion people will need access to improved sanitation over the period 2005-2015 (United Nations Department for Economic and Social Affairs [UNDESA], 2007; World Water Development Report [WWDR], 2009). The delivery of sustainable waste management services in developing countries has therefore become an issue of grave global concern.

In line with the United Nations' blueprint for sustainable development, Agenda 21 (United Nations Division for Sustainable Development [UNDSD], 1992), which recommends support for developing countries as a step towards the agenda, a number of developing countries have requested the collaboration of external support agencies in improving environmental sanitation services delivery. To achieve the MDGs in the poorest and most disadvantaged countries, the United Nations (UN) recognises the need for developed countries to deliver on longstanding commitments to achieve an official development assistance (ODA) target of 0.7 per cent of gross national income (GNI) by 2015 (UNDESA, 2007). However, many project interventions in direct waste management service improvement by external support agencies have failed to provide lasting positive impacts on the state of environmental sanitation in the recipient developing countries (Menegat, 2002). Many have failed to continue activities after the external support agencies ceased their support (Ogawa, 2000; Pronk, 2001). Several authorities have pointed to the strength of the institutional structures and arrangements as a key underpinning factor to sustainable development in water and sanitation (Department for International Development [DFID], 1998; World Bank, 2000; Antipolis, 2000; Ogawa; 2000).

According to the DFID, the central lesson learned from the strong emphasis laid on the construction of new facilities in the 1980s and 90s is that, simply building new facilities does little to help the poor. "Projects that end with the construction phase inevitably fall into disrepair and disuse unless hardware installation is fully integrated with properly planned and implemented arrangements for the long-term operation, maintenance and financing of an improved service" (DFID, 1998: 118). This, the DFID explains, is because poor management of facilities leads to declining service levels, which in turn reduce the chances of good cost recovery in terms of both willingness-to-charge and willingness-to-pay (Heller et al, 2003; Wunder, 2005; Pagiola et al, 2005). Invariably, governments and municipal authorities are unable to ensure efficient operation and maintenance, let alone, to ensure that investment in the sector keeps pace with the increasing demand.

1.2 The problem

In spite of the inescapable connection between institutions and the attainment of sustainability in waste management, sections of the engineering and technical professionals commonly involved in the practice of waste management tend to possess a distorted view of the concept of institutions and are therefore susceptible to over simplifying the concept or ignore sensitive aspects which eventually affect the viability of the whole institutional framework. A common misconception is the equation of institutions to organisations whereas, in reality, the latter is only an aspect of the former. Beside this, the consideration of institutional issues has often been limited to the formal segment, oblivious of the crippling effect which informal institutions exert on the whole institutional framework. Consequently, efforts at developing institutions to pursue sustainable waste management tend to produce institutional interventions which fail to adequately respond to the root factors that determine whether or not service delivery will be sustainable. With reference to developing countries, in particular, the questions this chapter seeks to answer relate to what basic understanding of the concept of institutions technical waste management professionals need to guide them in the development of responsive institutional interventions required to ensure sustainability in waste management services, and the relationships between the institutional matrix and the prospects of sustainability in waste management.

1.3 Chapter objective and approach

The purpose of this chapter is to analyse the institutional matrix within which waste management services are rendered to explain key institutional issues which underpin the chances of attaining sustainability in service delivery – with special emphasis on developing countries. This is intended to equip technical waste management professionals with a basic understanding of the concept of institutions required to inform a holistic approach to the task of institutional development in the sanitation sector, and to make them appreciate how institutional issues affect sustainability in all of its dimensions.

The arguments and concepts presented in this chapter are drawn from literature and the knowledge and experiences of the authors. Reference is often made to cases in Ghana where a practical illustration is considered useful.

2. Understanding institutions

A successful application of institutional development as a tool for attaining sustainability in waste management service delivery requires a proper understanding of the concept of institutions due to its complexity, especially, in the developing world. There is no doubt that in everyday language the term *'institution'* is used interchangeably with the term *'organisation'* as in such uses as *'academic institutions'*, *'financial institutions'* and the likes. Nevertheless, a clear distinction is made between organisations and institutions in literature (Bandaragoda, 2000; Young, 2002; Alaerts, 1997; Uphoff, 1986; Pai & Sharma, 2005). The other dimension that exists in the use of the word is rather sociological and not as yet understood by sections of the engineering and technical professionals frequently encountered in the environmental sanitation sector. This dimension reflects in such uses as *'the institution of marriage'*, *'the chieftaincy institution'* and *'the institution of priesthood'*, which are not necessarily organisations.

According to North (1990: 4), institutions are "formal rules, informal constraints - norms of behaviour, conventions, and self imposed codes of conduct - and their enforcement characteristics". The DFID (1998: 154) elaborates North's definition. According to the DFID, while an institution may be defined as "a set of constraints and humanly devised rules which influence and shape the interaction and behaviour among groups and individuals" – akin to North's definition – it may also refer to an individual organisation, i.e. "an individual body with an explicit structure and hierarchy of authority and the formal allocation of tasks and responsibilities". These bodies with an explicit structure and hierarchy of authority – i.e. organisations – enforce rules, norms, conventions and codes whether formal or informal. The emphasis, however, is that the term 'institution', does not necessarily refer to an organisation but, first and foremost, refers to a set of rules and arrangements existing in society to influence and shape interaction and behaviour among groups and individuals (North, 1990; Kingston & Caballero, 2008). For the avoidance of confusion, DFID (2003) distinguishes between institutions and organisations as suggested by North: it refers to institutions as 'the rules-of-the-game' and organisations as 'how we structure ourselves to play', adding that the key distinction between institutions and organisations is that between rules and players.

From this point onwards in this chapter, a conscious effort is made not to use the two bed-fellow terminologies interchangeably. 'Institution' shall be used for the 'rules-of-the-game' and 'organisation' for bodies or 'the players' unless where 'institution' is used to represent the totality of the 'rules' and the 'players' as in cases where mention is made of the 'institutional structure' or 'institutional framework' which encompasses both the rules and the players.

It is worth mentioning that the isolation of one interpretation from the other is practically impossible as the existence of one often elicits the other. For instance, the promulgation of "a set of rules to influence and shape the interaction and behaviour among groups and individuals" (DFID, 1998: 154) often leads to the creation of bodies or organizations – whether in the form of committees, commissions or agencies – to implement, administer or enforce the set rules. On the other hand, every organisation has its own rules which define its "hierarchy of authority and the formal allocation of tasks and responsibilities" (DFID, 1998: 154). Thus, institutions are implicit in organisations and the vice versa. In Ghana, for example, a policy to separate rural water supply and sanitation from urban water supply and sanitation service delivery led to the creation of the Community Water and Sanitation Agency in 1998. Meanwhile, it took an act of Parliament to establish the Agency. Thus the new policy or institutional arrangement necessitated the creation of an organisation whose establishment in turn called for further legislation from which it derives its "hierarchy of authority and the formal allocation of tasks and responsibilities".

3. Institutional matrix for the sanitation sector

The application of a holistic view of the concept of institutions to the sanitation sector reveals a two-by-two *institutional matrix* with the two columns representing institutions and organisations and the two rows partitioning them (institutions and organisations) into formal and informal hemispheres as shown in Table 1.

	Institutions	Organisations
Formal	• Policies • Laws • Regulations • Guidelines • Codes • Standards, etc.	• Government Ministries, Departments and Agencies • Municipal authorities • Private sector organisations • Non-governmental organisations (NGOs) • External support agencies, etc.
Informal	• Customs • Beliefs • Norms • Values • Historical experiences • Practices • Standards of honesty, etc.	• Traditional leaders • Pressure groups • Clans and family gates • Religious groups • Social clubs • Community watchdog committees • Community-based organisations, etc.

Table 1. Institutional matrix for the sanitation sector

Thus four segments of the matrix can be distinguished, namely formal institutions, informal institutions, formal organisations and informal organisations, all of which play crucial and interrelated roles to ensure the overall viability and sustainability of the institutional framework. It must be mentioned, however, that the structure of the matrix is not peculiar to waste management or the sanitation sector *per se.* Almost all sectors of a nation's economy is characterised by institutions and organisations – both formal and informal.

3.1 Formal institutions

Formal institutions set the tone for the sector. They are the 'formal rules' in North's definition, which "influence and shape interaction and behaviour" (Hearne, 2004; Kingston & Caballero, 2008) among sector stakeholders including service providers, users and government itself. They come in the form of laws, policies, regulations, guidelines, codes and standards etc. They also include international treaties and protocols to which the national government is a signatory. National, state or regional and municipal authorities usually promulgate them. As the sceptre of governance, their presence or absence is the most critical factor that determines the level of orderliness or chaos that can be expected to exist in the waste management industry and the sanitation sector in general.

While each of the segments of the matrix has a potential influence over the others, formal institutions are the most powerful. This can be explained by the fact that formal institutions decide which informal institutions or constraints can be adopted, tolerated or outlawed since governments can restrain by law what is culturally acceptable or technically feasible. For instance, many developing countries have enacted laws to ban the use of the pan or bucket latrine, which has been practised by some communities for several years. Consequently, this option for excreta disposal has given way to better practices and the organisational framework, both formal and informal, within which it was carried out has collapsed. Again, many nations have formally outlawed female genital mutilation, which has been practised by some cultures for centuries. Such is the strength of formal institutions.

3.2 Informal institutions

Informal institutions can be best described as the *unwritten rules* which govern behaviour (Helmke & Levitsky, 2004). These are the unofficial arrangements, which exist in society or organisations and influence the standard of acceptable or objectionable conduct. They often manifest themselves in traditions and cultural practices that are performed by the members of a society. They have been in existence for centuries and are a reflection of the deep-seated traditional value system of people and can be reflected in the formal institutional framework of a society (e.g. constitutions, laws, legal mechanisms) (Helmke & Levitsky, 2004).

In the environmental sanitation sector, informal institutions and constraints are major determinants of the commitments of various stakeholders to the enforcement of and compliance with formal institutions (Vogler, 2003). They influence such critical factors as attitudes to personal hygiene, waste disposal practices, willingness to pay for services, commitments to public interests and law enforcement, respect for sanitation professionals, etc. The impact of informal institutions and constraints on the sanitation sector is generally more pronounced in developing countries than in the developed world, where formal institutions are much better developed.

While some informal institutions tend to promote best environmental sanitation practices, others have a tendency to interfere with them (Alaerts, 1997). Traditional or cultural institutions, which uphold sound environmental practices, are to be harnessed and integrated into the local institutional arrangements. Where informal institutions conflict with best practices, formal institutions are used to constrain or outlaw them, but not just by the might of laws and regulations. This is because changing informal institutions require much tact, intensive education, stakeholder participation, dialogue and incentives (Hall & Thelen, 2005) because institutions have a degree of permanence and are relatively stable. It is also as a result of the fact that institutional change is viewed as a centralized, collective-choice process (Kingston & Caballero, 2008; Kantor, 1998). In this process, it is argued that "rules are explicitly specified by a collective political entity, such as the community or the state, and individuals and organisations engage in collective action, conflict and bargaining to try to change these rules for their own benefit" (Kingston & Caballero, 2008: 4).

3.3 Formal organisations

Organisations are groups of individuals engaged in purposive activity (North 1990; Saleth, 2006). Described as the 'players' (DFID 1998, DFID 2003, North 1990), organisations, in general, are the primary custodians of institutions as well as the wheels on which they (institutions) are run.

Formal organisations are those with some form of officially recognised authority. They are material entities possessing offices, personnel, equipment, budgets, and legal personality (Bandaragoda, 2000). They are bodies with explicit structure and hierarchy of authority. Government ministries, departments and agencies, municipal authorities, private companies, non-governmental organisations (NGOs), external support agencies, etc, are among the formal organisations which play various roles in the delivery of waste management services within a framework defined by formal institutions. Thus, formal organisations are subject to formal institutions which may be promulgated by the self-same organisation. This illustrates the paradox of institutional-organisational relationships: institutions are evolved by bodies or organisations – be they state departments, ministries, commissions or the parliament – but all bodies or organisations are themselves built on and governed by institutions.

Direct waste management service delivery has often been a shared responsibility between state and private organisations, engendering a wide range of public-private partnerships. Water and Sanitation for Health [WASH] (1991) notes that the pressures to become more efficient and effective are changing the role of the government from that of a provider to a promoter and regulator. For instance, Obeng *et al* (2009) studied the impact of Ghana's Environmental Sanitation Policy on the institutional structures for solid waste management in Kumasi, the nation's second largest city. The study found that the major change that had occurred in the organisational structure for the management of solid waste in the city since the inception of the policy in 1999 was the involvement of the private sector in service delivery under the supervision and monitoring of the Waste Management Department (WMD) of the Metropolitan Assembly (see also Cook & Ayee, 2006).

3.4 Informal organisations

Informal organisations are groups with some common interests or aspirations who may not be officially established or registered by the national or local government but can be recognised as stakeholders in the delivery of waste management services due to their potential to affect the chances of successful service delivery positively or negatively. They include community-based organisations, pressure groups, opinion leaders, traditional leaders, gender groups, local religious bodies, etc.

The potential of informal organisations to affect the chances of sustainable service delivery has gained much attention in recent times, leading to the high emphasis that is currently laid on effective community participation in service delivery in developing countries (Menegat, 2002). Stakeholder analysis for community participation helps to identify all interest groups in the community, assess the conditions for their involvement in order to attract each group to fully participate in identification, planning and implementation of sanitation and waste management intervention programmes at the community level.

4. Institutions and sustainability in waste management

4.1 Overview of the concept of sustainability

The concept of sustainability, which literally refers to "the ability to sustain, or a state that can be maintained at a certain level" (Kajikawa, 2008: 218), arose out of the belief that the growing population of the world, with the attendant pressure on natural resources, poses a threat to our survival on the earth. Back in 1798, Thomas Malthus argued that unchecked population growth follows a geometric order while subsistence for man increases arithmetically. Therefore, in the opinion of Malthus, if human populations and consumptions are not controlled, the earth would run out of its resources at some point in time (Malthus, 1798 as cited in Rogers *et al*, 2008).

This concept, which currently occupies a central position in all developmental issues, initially attracted the attention of the international community in 1972 when the United Nations Conference on the Human Environment in Stockholm first explored the relationship between the quality of life and that of the environment (Rogers *et al,* 2008). As the interaction between human populations and the environment are essentially the outcome of our quest for development, the term 'sustainability' became more associated with the term 'development' than any other. This has led to the frequent use of the phrase 'sustainable development' which was first defined by the World Commission on Environment and Development (WCED) as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED, 1987: 54).

The concept of sustainability refers to a "dynamic condition of complex systems, particularly the biosphere of earth and the human socioeconomic systems within it" (Heintz, 2004: 6). The concept draws on the fact that societal development cannot be viewed without considering its natural prerequisites (United Nations Educational, Scientific and Cultural Organisation [UNESCO], 1996). Sustainable development then refers to a pattern of resource utilisation that seeks to meet human needs while preserving the environment so that these needs can be met in the present as well as in the future (Valverde, 2008). The term has come to encompass the economic, environmental and social realms (Hasna, 2007). It also includes the bio-chemical and physical dimensions (Gupta and van der Zaag, 2008).

This has informed the views of sustainability as the 'triangular view' which treats sustainability as being triple-dimensional, with three components addressing the need to sustain the environment, economy and society (Kajikawa, 2008; Rogers *et al,* 2008). Thus, Kajikawa (2008) describes the triangular view as including the three-pillar model in which the three pillars refer to the economy, the environment, and society (Kastenhofer and Rammel, 2005) and the triple-bottom-line model (People, Planet, Profit) or P3 (People, Prosperity, and the Planet) (Zimmerman, 2005). It can be argued then that a sustainable system or development is one which satisfies environmental sustainability (the sustainability of the planet), economic sustainability (the sustainability of prosperity or profit) and social sustainability (the sustainability of the values and cultures of people). Thus, a sustainable waste management system is one oriented at attaining all three components of sustainability: environmental, economic and social. It is important that each of the three components is given equal attention and priority in order to ensure sustainable outcomes (Rogers *et al,* 2008).

4.2 Institutions and environmental sustainability in waste management

In simple terms, environmental sustainability implies that human developments or activities such as waste disposal should not hinder the ability of biological and physical systems to maintain their ecological resilience or robustness (Rogers *et al,* 2008). That is, levels of harvest should be maintained within the capacity of the ecosystem (Kajikawa, 2008). In waste management, environmental sustainability implies that, the rates of deposition of pollutants should be maintained within the rate at which the ecosystem can safely absorb or convert those pollutants to some other useful or harmless substances. Thus, the environment should only be used as a "waste sink" "on the basis that waste disposal rates should not exceed rates of managed or natural assimilative capacity of the ecosystem" (Pearce, 1988 as cited in Rogers *et al*, 2008: 43). In the design of sanitary landfills, for example, the provision of a lining material and physical installations to prevent leachate from reaching ground water resources is intended at enhancing the environmental sustainability of that disposal option.

Institutions play a vital role in ensuring environmental sustainability in waste management. This vital role becomes apparent as one reflects on the determinants of environmental protection such as:

- legislation and regulation to restrain or outlaw waste disposal practices which adversely affect the environment;
- monitoring and enforcement to detect and punish environmental abuse and malpractice;
- research to determine the capacity of the environment that can safely absorb different types of wastes and the technology options by which waste managers can make optimum use of this capacity.

All aspects of the institutional matrix contribute immensely to ensure that the determinants of environmental sustainability, including but not limited to those mentioned above, are in existence.

4.2.1 Formal institutions and environmental sustainability

Formal institutions in the form of laws, regulations, policies, standards and guidelines often take the lead in the pursuit of environmental sustainability. Examples around the world include:

- national laws such as the Resource Conservation and Recovery Act (1976) of the United States (United States Congress [U.S.C.], 1976) and the Hazardous Waste (England and Wales) Regulations (2005) of the United Kingdom (Statutory Instruments, 2005);
- regional directives such as those of the European Union, including the Regulation (1272/2008) on classification, labelling and packaging (CLP) of chemicals (European Union, 2008) and the Directive (2002/96/EC) on waste electrical and electronic equipment (WEEE) (European Union, 2003); and
- international conventions such as the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal (Basel Convention, 2010).

Arguably, the worst form of failure in securing the sustainability of the environment is the absence of formal institutions, at least on paper, to prohibit certain forms of actions and inactions which threaten the survival of vital ecosystems. Hence, the analysis and diagnosis of the overall institutional framework is recognised as the first step in the institutional development process in the water and sanitation sector (DFID, 2003) and, for that matter, in the field of waste management. One of the major factors which account for the differences in waste management in the developed and the developing world lies in the existence of formal institutions. For example, with respect to electronic waste, Zhao *et al, (2009)* note that developing countries have no laws or relaxed legislations.

4.2.2 Formal organisations and environmental sustainability

As custodians of formal institutions, formal organisations – including legislative assemblies, environmental protection and regulatory agencies, local authorities and waste management companies – are not only involved in the promulgation of formal institutions but also see to their implementation and enforcement. Research and academic 'institutions', as they are commonly referred to, are among the formal organisations which work hand in hand with waste management practitioners in the development of environmentally sustainable technologies.

The commitment and capacities of formal organisations existing in a nation are key determinants of the kind of formal institutions which would be developed and the extent to which they (formal institutions) are implemented and enforced or rather remain dormant. Private companies, for instance, seek to minimise operational costs in order to maximise profits (Coad, 2005; Cointreau-Levine, 2000) and would naturally crave the absence or relaxation of formal institutions which impose strict waste disposal regulations that have implications for operational costs. Therefore, it is always important to have a strong regulatory capacity within the public sector to regulate and monitor the private sector. A low level material capacity combined with a shortage of skilled staff and training leads to inefficient performance (Antipolis, 2000), and this is another key factor which distinguishes waste management in developing countries from that of developed countries.

4.2.3 Informal institutions and environmental sustainability

Informal institutions – including traditions, customs, beliefs, values and attitudes – play vital roles in waste management at the community level. In rural areas of developing countries, especially, where formal education is usually low and formal institutions either unknown or ignored, traditional authorities tend to apply traditional laws and customs to protect the local environment.

Nevertheless, it is not uncommon for informal institutions to conflict with best environmental practices. It is therefore important for waste management practitioners to understand the informal institutions existing in a community while selecting technologies aimed at protecting the environment in those communities. It is also imperative to incorporate informal institutions which promote good waste disposal practices in formal institutions while making use of the latter to outlaw the former, where they are found environmentally unfriendly. However, such a move should be accompanied with intensive education to convince traditional people of the need to abandon an age-old tradition in the light of new knowledge.

In the formal sector, popular opinion and values could also compromise the role of monitoring and enforcement in environmental protection. For example, when the values system makes it attractive for the enforcement official to connive with the waste generator or Collection Company to violate existing waste disposal regulations, environmental sustainability is compromised. The existence of monitoring and enforcement mechanisms is heavily predicated on the assumption that the officer-in-charge is not corruptible but, in some cases, that may not be true.

4.2.4 Informal organisations and environmental sustainability

Informal organisations, like their formal counterparts, are the custodians of informal institutions and play a role in applying them to protect the environment. Again in developing countries, where the low capacity of formal regulatory and policing organisations does not allow a close monitoring of communities and private organisations, informal organisations such as community watchdog committees and gender groups could be empowered to monitor compliance to formal and positive informal institutions at the community level. Examples exist in Ghana, where Water and Sanitation Development Boards (WSDBs) exist in small towns (for piped water systems) and WATSAN (water and sanitation) committees in small communities and villages (for single source water systems) to extend the powers of the local authority (District Assembly) closer to the communities to, among other responsibilities, ensure a safe environment for all community members.
Traditional authorities may be empowered to impose sanctions on offending community members who engage in waste disposal practices that are detrimental to the sustainability of the environment.

4.3 Institutions and economic sustainability of waste management

The economic sustainability component cautions against deriving today's wealth or achieving some other environmental or social benefits in a manner that diminishes the overall stock of capital or resources including natural resources (Rogers *et al,* 2008; Valverde, 2008). According to the World Bank, the pursuit of sustainable development should base developmental and environmental policies on a number of factors including a comparison of costs and benefits (World Bank, 1992 as cited in Rogers *et al,* 2008). In practical terms, waste management should be done in a manner that can be justified when the overall benefits – including the estimated economic value of environmental protection and resource recovery – are compared with the economic cost of the service. In solid waste management, for instance, the desire for economic sustainability justifies the practice of resource recovery, recycling and reuse, which reduce the quantity of wastes to be eventually disposed of in sanitary landfills. By these practices, the costs of collection and transportation to final disposal sites, as well as the "consumption" of land for landfilling, are considerably reduced. The role of institutions in ensuring economic sustainability resounds in such economic issues as availability of capital for infrastructure development, recovery of costs and operational efficiency.

4.3.1 Formal institutions and economic sustainability

Formal institutions determine minimum service standards and requirements for waste disposal by corporate and individual citizens. These in turn determine the cost of service delivery. Besides, formal institutions determine whether or not:

- waste management services remain a statutory responsibility of the municipal authority, may involve the private sector or must certainly be delegated to the private sector;
- service should be provided as a social service (public good) or as an economic enterprise; and
- costs should be recovered in full or partially.

Answers to these questions and, for that matter, the formal institutional framework are decisive because the World Bank (2000) notes that an acceptable level of service for waste management depends critically on a well planned management, operating within an enabling institutional framework capable of generating the financial resources required to meet operating, maintenance and investment cost.

An example of the relationship between formal institutions and economic sustainability can be found in Obeng *et al* (2009) relating to solid waste management in Kumasi as cited earlier. Prior to the inception of Ghana's Environmental Sanitation Policy in May 1999, solid waste collection services were provided by the Kumasi Metropolitan Assembly as a social service without any charges to beneficiaries. However, the policy introduced private sector participation as one of its key strategies towards cost recovery (Ministry of Local Government and Rural Development [MLGRD], 1999). Private companies were contracted to collect waste from communal storage points and also franchised to provide house-to-house collection services to households within various zones demarcated throughout the metropolis. While waste collection from communal storage points was paid for by the central government, the cost of house-to-house collection was borne by the individual households without any subsidy from the government. Thus, the growth of house-to-house collection services led to cost recovery or, better still, cost savings on communal waste collection, since wastes collected under house-to-house service would have otherwise been deposited in communal bins. The study found that the amount recovered from house-to-house collection services, as a percentage of the expenditure of the Metropolitan Assembly's Waste Management Department (WMD), increased from 26.5% in 2001 to 68.6% in 2004, as shown in Figure 1.

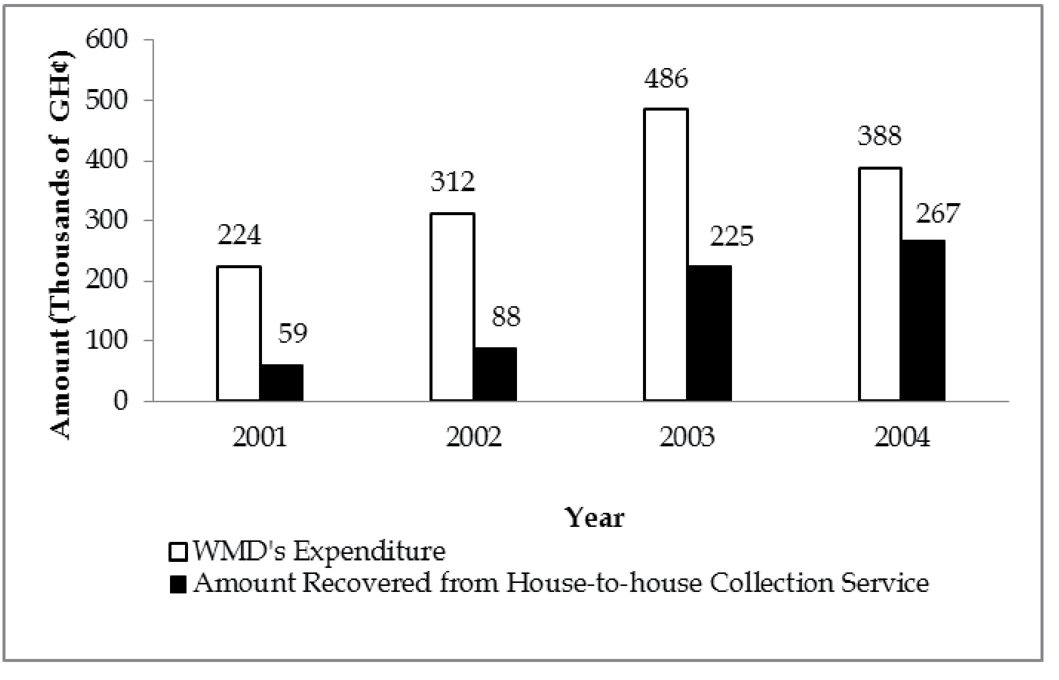

Fig. 1. Amounts recovered from solid waste collection in Kumasi after Ghana's Environmental Sanitation Policy of 1999 introduced private sector participation (*Source: Obeng et al [2009]*)

4.3.2 Formal organisations and economic sustainability

The influence of formal organisations on the economic sustainability of waste management is demonstrated by the popular debate over whether to leave waste collection services in the hands of public or private organisations. It is certain that waste management services cannot be economically sustainable unless some key organisational factors exist. These include:

- willingness and ability to invest in the acquisition of adequate equipment to provide the level of service which justifies service providers' willingness to charge and elicits beneficiaries' willingness to pay;
- a good commercial orientation and operational efficiency, marked by use of optimal workforce and low cost of operation, to generate profit or recover cost without necessarily charging exorbitant tariffs;
- a healthy competition among service providers

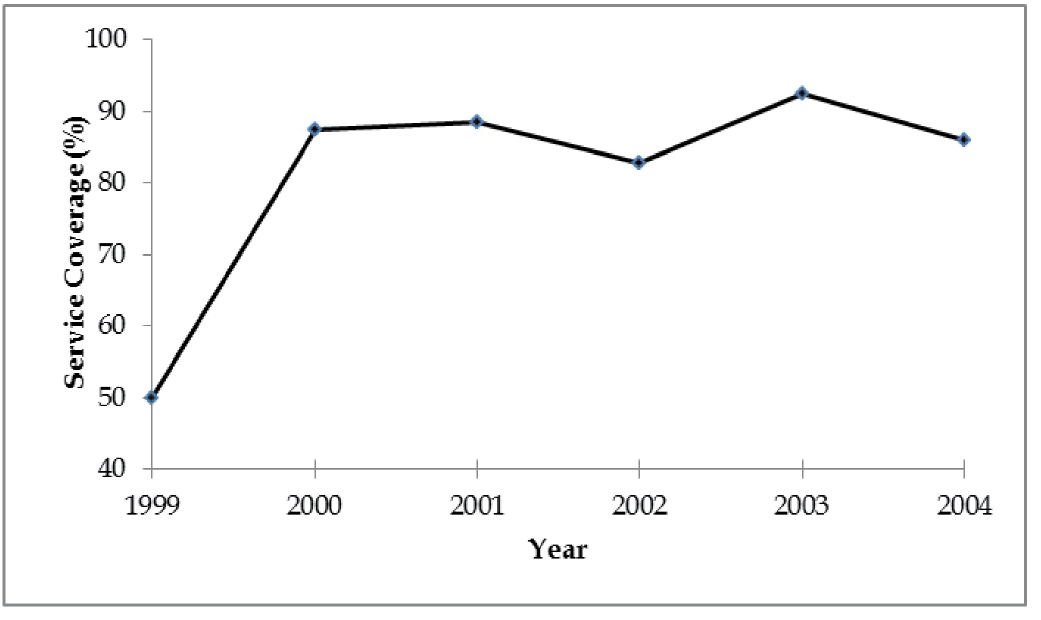

Fig. 2. Solid waste collection coverage in Kumasi after introduction of private sector participation (*Source: Obeng et al [2009]*)

The existence of the above requirements for success tends to make the above mentioned debate end in favour of the private sector. For instance, the study by Obeng *et al* (2009) found out that the involvement of the private sector in Kumasi had led to an increase in service coverage in terms of the proportion of the estimated waste generation that was collected by the private companies. As shown in Figure 2, the annual average prior to private sector participation was about 50%. However, upon the introduction of private sector participation, the annual average ranged between 82.8% and 92.5% between 2000 and 2004. The trend was attributed to the fact that the private companies had come along with equipment which the Waste Management Department of the Kumasi Metropolitan Assembly could not acquire while it provided the service directly.

It must however be mentioned that, the introduction of private sector participation *per se* is no panacea to economic sustainability. Without adequate measures to regulate profit-seeking private companies, they tend to exploit the public and cut costs at the expense of quality service. If the external costs of compromised service quality to beneficiaries and the environment are internalised, the service may actually be found to be economically unsustainable.

4.3.3 Informal institutions and economic sustainability

Informal institutions play a significant role in the economic sustainability of waste management in rural areas of developing countries in particular. They determine the values, perceptions and attitudes of waste management service practitioners and beneficiaries towards the economic aspects of waste management, as well as the unofficial arrangements which may exist at the community level to respond to the requirements of economic sustainability.

Where traditional values place a high priority on personal and communal hygiene and institutes penalties for offenders, it is easier to convince service beneficiaries to make financial contributions towards service delivery and, hence, improve the chances of cost recovery. Besides, traditional laws may be used to control indiscriminate disposal and consequently minimise the cost of cleansing activities and the external cost of environmental pollution.

4.3.4 Informal organisations and economic sustainability

Informal organisations, usually at the community level, affect economic sustainability in a number of ways especially in developing countries. They offer informal structures by which informal institutions are applied to enhance economic sustainability. For instance, in rural Ghana, WATSAN committees mobilise community members to provide direct environmental management services such as drain cleansing and public latrine management in order to minimise or avoid the cost of hiring hands for the service. Also during the construction of waste management infrastructure, volunteer, youth and gender groups in the community may contribute labour to minimise the cost of the project. In that case, the poor can also contribute to the economic sustainability of services and projects without making any financial contributions. On the other hand, those who can afford financial contributions are allowed to do so in lieu of direct involvement while their contributions are used to engage the jobless to provide the service.

Co-operative groups also help each other to acquire household facilities by making regular contributions into a mutual fund. Community members who are not capable of making a one-off payment for the acquisition of such facilities like domestic toilets take advantage of such schemes to acquire them and pay for it over a conveniently long period of time.

4.4 Institutions and social sustainability of waste management

Social sustainability reflects the extent to which the stability of social and cultural systems is unaltered by the pursuit of one development agenda or the other. Thus the quality of lifestyles and the values of a society should not be compromised in a bid to satisfy some other environmental or economic aspirations. If social sustainability is violated, the reactions of citizens, communities and governments toward an otherwise well-intended environmental or developmental initiative are negative and uncooperative (Cox and Ziv, 2005). This is supported by the maxim that says "people do not resist change; they only resist being changed". Thus a socially sustainable waste management scheme is one that is packaged in a manner that demands minimal inconvenient change in lifestyles or values of stakeholders or rather introduces a radical change through effective social marketing strategies that make the stakeholders perceive themselves to be better off with the change.

Where social sustainability is not achieved, people prefer to continue with their 'own way of life' rather than to adopt a new technology or facility which promises to offer environmental sustainability or even economic sustainability at the expense of some traditional values. This situation arises where the new technology calls for the abandonment of one traditional practice or the other. For instance, house-to-house refuse collection services, which have helped to reduce backyard dumping of refuse and led to cost recovery in many Ghanaian cities, was initially not patronised by some tradition-oriented people who could not sacrifice the traditional practice of emptying the waste bin every morning and/or evening for the weekly collection service offered by private companies. Meanwhile, it makes economic sense to provide a household with a large bin for a week-long storage so that the collection crew move into a particular neighbourhood weekly rather than daily, as some conservative traditionalist would prefer.

Harmonisation of formal and informal institutions and collaboration between formal and informal organisations are ways by which institutions could be used to deal with such situations and enhance social sustainability in waste management. For instance, in Ghana, the National Community Water and Sanitation Programme (NCWSP) prepared by the Community Water and Sanitation Agency (CWSA) seeks to address sustainability issues in rural water supply and sanitation. The programme adopts the community ownership and management (COM) approach to avoid problems of sustainability, especially social and economic sustainability. Under the programme, the CWSA only plays the role of facilitators while the community exercises the freedom to select technology options, under the guidance of the CWSA's technical team, and elects representatives to constitute a WATSAN committee or WSDB to manage and operate the facilities. Thus, there is opportunity to blend customs with best environmental and technical practices to the acceptance of members of the community.

This collaboration between the CWSA – a formal organisation – and the WATSAN committee or WSDB – informal organisations – is possible because the formal institutional framework allows it. The District Assemblies have bye-laws from which the WATSAN committees and the WSDBs derive their authority.

5. Conclusion

From a holistic perspective, institutions are not just about organisations but, first and foremost, the arrangements and rules, which exist in society to control behaviour and interactions among individuals and groups, both formal and informal. The institutional matrix for waste management should be viewed as consisting of four interrelated components, namely formal institutions, informal institutions, formal organisations and informal organisations.

All aspects of the institutional matrix play crucial roles to ensure sustainability in the delivery of waste management services and account for the differences observed in the quality and sustainability of services between developing countries and their developed counterparts. It is also noted that, the effects of informal institutions on the whole institutional matrix is more pronounced in developing countries than in developed countries.

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Waste Management Facility Siting and Social Conflicts – the Case of Hungary

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1. Introduction

Frank Popper was the first to have used the concept of LULU (Locally Unwanted Land Uses) in 1981. LULU may refer to low-cost housing, power plants, airports, wastewater treatment plants, prisons, open-cast mines, energy supply lines, motorways, dams, oil refineries, railway lines, landfills, cemeteries, amusement parks or pubs or military facilities (Popper, 1981). Almost every major regional development project (petrol station, car repair shop, motel, parking house, rent-a-car company etc.) behaves like a LULU, and often even facilities which at first sight seem to be desired by the community (office building, residential park, luxury hotel, hospital, assembly shop, port etc.) come to the fate of LULUs. It would be very difficult to describe by one word why these facilities sometimes provoke such extraordinary resistance. Fear from physical injuries may be a motive, the same as concern about the stigmatisation of the host settlement and the consequent drop in real estate prices. Some negative impacts are certain to occur (e.g. increase in air pollution and noise load near a newly completed motorway section), whereas others have a very low occurrence probability (such as the leakage of a nuclear waste repository, for example). The negative health or economic impacts may be accompanied by negative social impacts, such as the erosion of the social networks or the often irreversible alteration of the local cultures (Lesbirel, 2003). In the context of an international comparative survey of the motives of protest against the siting of low-level nuclear waste repositories, Anna Vári and her fellow researchers came to the conclusion that concerns about undesirable facilities typically fall into five categories: the opponents of the repositories expressed health and safety, economic, environmental and social as well as technical and decision-making-related concerns (Vári et al., 1991).

In addition to LULU, another commonly used acronym in connection with the siting of facilities of the above type is NIMBY, i.e. not in my backyard. For the purpose of completeness, let me mention that in addition to the two well-known acronyms (LULU and NIMBY), new ones have also appeared in the technical literature and the media, such as NOPE (Not on Planet Earth) or BANANA (Build absolutely nothing anywhere near anybody). The NIMBY phenomenon carries an important additional meaning relative to LULU: it is used mainly to denote investments considered reasonable even by the opponents of the facility (a refuse incinerator, for example, which is necessary at national or regional level, and the absence of which could threaten waste management), who question only why it needs to be built in their backyards and not elsewhere (Sjöberg–Drottz-Sjöberg, 2001).

In this chapter the social conflicts around the siting of Hungarian waste management facilities will be introduced and discussed. In the following section a general overview on the environmental conflicts of the transition period is given, and then in section 3 the research methodology is presented. Section 4, 5 and 6 deal with concrete mini case studies on siting conflicts. In section 7 the case studies are discussed, while section 8 contains some conclusions.

2. The transition and environmental conflicts in Hungary

The communist regimes of Central and Eastern Europe collapsed in 1989 enabling the set up of new democracies bringing the right for free speech, free elections, market economy, etc. to these countries. Post-communist societies had to learn to exploit the opportunities inherent in the newly established political and economic systems. Among many revelations, people began to understand what participation in local politics meant, how they could articulate their opinion in the local political arena, or how they were able to veto certain undesired decisions in regional and municipal level. Although transition in Central and Eastern European countries happened with different pace and intensity, Hungary definitely was one of the countries that adopted the new institutions the most rapidly at the beginning of the political transformation. The history of environmental decision making is an emblematic example of this learning process.

2.1 Siting conflicts as a form of anti-communist movements

Hungarian research focusing on environmental decisions began in the 1980s. The most important case of the era was the conflict concerning the construction of the Bős-Nagymarosi Dam, but the siting of the hazardous waste incinerator in Dorog also provoked many disputes (Faragó et al., 1989). These cases highlighted an important aspect of the situation in Hungary: they culminated at a historical moment which defined their further fate. The dam was a symbol of political power threading through everything of the totalitarian party regime (Fleischer, 1993). Protest against the dam was obviously a way to demonstrate against the regime, and hence the conflict acquired a connotation that was different from that of the typical Western European and North American conflicts. By the end of the eighties, it had become impossible to prevent the population from expressing its opinion on environmental decisions. One of the most important features of the siting conflicts at the time of transition, namely their dynamically changing institutional, political and social environment (something that existed hardly or not at all in Western European research) may reinforce our belief that the social institution systems (or their absence) have a major influence on the course of progress of social conflicts of this type.

The case of Bős-Nagymaros, however, highlighted many other factors as well. Firstly, it turned out that the tug of war of the actors of the central political power often manifests itself in specific cases and overrides expert considerations. Secondly, since the different positions ought to have been reconciled with the contribution of Czechoslovakia and later on Slovakia, the conflict, quite severe anyway, acquired an international dimension. The case of the dam shed further shadows on the far-from-cloudless relationship of the two countries, laden with conflicts historically. The national governments often used the case of the Dam to achieve their own home policy targets, without making a real effort to come to a solution. Decades later a similar dilemma recurred in the form of the conflict provoked by the siting of the Rosia Montana (Romania) gold mine, when the Hungarian state took action as stakeholder of the case.[1]

The siting of the Dorog hazardous waste incinerator was another problem given excessive media coverage in the eighties. The case was present throughout the decade, and it differed from that of the Dam in that it displayed problems similar to the ones we encountered in the Western European and North-American literature. The plan of Kőbánya Pharmaceutical Factory to site a hazardous waste incinerator in Dorog, near Budapest, met with strong opposition there. There were already several polluting plants in the town (the pharmaceutical factory, a power plant and a briquette plant), and these imposed a significant burden on its environmental status. The residents were afraid that a new polluting factory would worsen the already unfavourable health conditions. The fact that the protesters held out despite the compensation packages being offered raises several questions. Knowledge commanded by the population and the experts, respectively, proved to be very different, the same as their points of reference. Distrust between the parties and their different value systems deepened the conflict even further (for a more detailed description of the case, see Faragó et al., 1989). The problems encountered in Dorog were obviously the same as those indicated by the relevant international literature (further pollution of areas exposed to environmental hazards already, and its moral aspects; reservations concerning compensation, and differences in risk perceptions).

Viktória Szirmai's study inquiring into the Hungarian circumstances (Szirmai, 1999) devotes an entire chapter to the "environmental social conflicts of the transition period", in which the author proposes a conflict typology, investigates the role of conflicts, and presents prevention options (Szirmai, 1999). Szirmai assigns environmental conflicts to five groups: conflicts to protect the values of the natural environment, protests against specific environmental damages, environmental conflicts related to waste management, urban development interventions, and infrastructure investments. In this study I focus on the environmental conflicts related to waste management, especially facility siting.

2.2 Facility siting and waste management

The transition period experienced a lot of siting conflicts in Hungary, many of them were related to landfills, dumps, and other waste management facilities. Around the date of EU accession the largest items of the Hungarian environmental state budget were spent on creation of new waste management facilities mostly with EU co-funding. Many of the landfills that were created before the transition period became obsolete and most of them were not in accordance with actual EU regulations therefore new facilities were needed.

Although industrial waste production has significantly dropped since 1990, the municipal waste generation has been still a crucial issue in the country (Pomázi, 2010). (Of course, municipal solid waste represents a lot smaller proportion than industrial waste production.) Municipal solid waste is mainly utilized in landfills (more than 80% of the waste is

[1] Since few scientific reflections exist so far on this case which met with such considerable press reaction, I shall refrain from its more detailed presentation and analysis here. However, it can be stated that several of the basic siting approaches appear in this case: investors and politicians arguing that the economically deprived region will prosper are opposed to the locals and the civil and other interest groups (the government included) concerned by the risks involved. As in the case of the Bős-Nagymaros Dam, party policy skirmishes, this time Romanian ones have made their impact on the development of the conflict.

transported to these facilities), some of them is incinerated (around 6%), and the rest is recycled or composted (see Figure 1.).

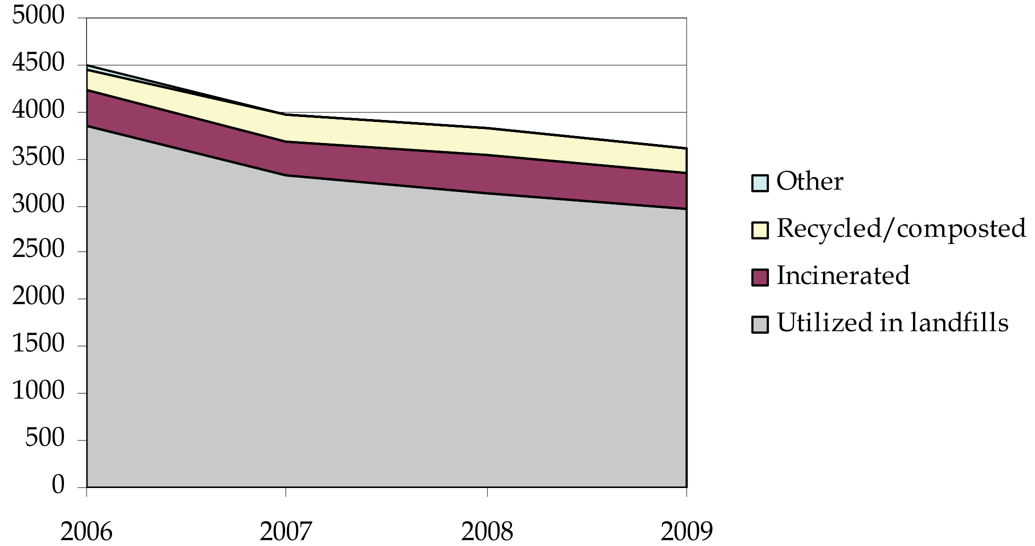

Fig. 1. Utilization of municipal solid waste (in thousand tons), source: HCSO

According to the relevant data, conflicts related to waste management issues mainly explode in minor settlements, with some exceptions (the cases that will be presented in this chapter are good illustrations for this phenomenon). In a previous study (Szántó, 2008) I pointed out that the settlements concerned are characterised by unemployment figures exceeding, occasionally excessively, the national average (the unemployment rate for example was thrice that of the national rate in Boda (County Baranya), considered a candidate for high level radioactive waste repository siting (see the case study later in this chapter), and in Liptód (in the same county), which rejected the siting of an envisaged waste disposal facility in 2000, it exceeded 30%). The higher rate of pension recipients in these settlements suggests a high rate of the elderly. If we compare that with the low amount of personal income tax payments, it is quite clear that the settlements where waste-related social conflicts have occurred are probably the worst off, and they are characterised by rather low income and excessively high unemployment rates.

One could assume that a settlement like that would jump at the chance of a new investment, but the experience of specific cases is that poor incomes and high-level unemployment are not enough to make the residents approve an investment almost automatically if the envisaged facility is thought to be highly problematic. Nevertheless, it is no accident that investors tend to look for such sites. They probably believe that the prospects of job creation, local tax revenues and other forms of compensation will make the local stakeholders accept the hazardous facilities more easily.

3. Methodology and data

Research methodologies based on case studies are especially popular in the area of siting decisions laden with environmental conflicts, not only in Hungary, but also in many other parts of the world where such research is carried out. This is no accident. Researchers meet with many viewpoints in the context of most siting decisions. There are many perspectives, and many actors with different motives, which makes it difficult to design and carry out a well-operationalised quantitative research project. Research is further aggravated by the difficulty of separating the phenomenon under study from its context (if that is possible at all), and this is another circumstance which may encourage the researcher to apply qualitative research methodologies (and in particular case study method) (Yin, 1994).

Case	Type of waste	Years
The North-east Pest County Waste Management program	solid municipal waste	2002-2004
North-Balaton Region landfill	solid municipal waste	2001-2010
Battery waste recycling facilities	hazardous waste	1985-2009
Alternative fuel in the cement factory	hazardous waste	2002-2005
Low-level nuclear waste repository	nuclear waste	1987-2009
High-level nuclear waste repository in Boda	nuclear waste	1995-

Table 1. Case studies on social conflicts

In this chapter six mini case studies are presented in order to highlight the most important factors of social conflicts around siting decisions (Table 1. contains the selected cases and their most important attributes). All case studies have occurred in the last two decades, but some of them have roots in the communist era. The case studies were elaborated by using data from primary and secondary sources. In two cases, I relied mostly on case study interviews (the waste battery processing plant and the Vác cement factory cases), while the other case studies are based on mainly secondary data sources: articles from the national and local press, and studies made by other researchers. Of course, these mini case studies are rather illustrations of the most important symptoms and underlying causes of the Hungarian siting conflicts, and they cannot be considered as a comprehensive description of the conflicts of the last two decades.

4. Solid waste facility siting

A series of events having “enjoyed” extensive media coverage in the last two decades, mostly problems associated with regional solid waste disposal facilities. Larger cities (Budapest and county capitals) have been facing with the dilemma of the growing need for larger solid waste disposal facilities that were able to handle the solid waste output of these communities. The concept of regional landfills was popularized by the European Union; hence more and more depositories were planned in the country. The deposition-oriented solid waste management in Hungary obviously has been triggering more and more conflicts; more focus on prevention of waste generation would be able to mitigate the intensity of disagreements in the future.

4.1 The North-east Pest county waste management program

The social conflicts around the plans of the North-east Pest County Regional Waste Disposal Facility might have received the greatest publicity amongst all cases (Kiss, 2005). Of course, it is not surprising. Pest County is located in the heart of Hungary, has many ties with the country's capital Budapest, and is one of the most developed regions of the country. The events of the siting 'saga' were published in major newspapers and a comprehensive case study was elaborated about the siting process (Kovács & Sándor, 2004). Since eight different villages were invited for accepting the waste management facility and all of them rejected the idea, the social conflicts were apparent even to the wider public. The rejections mostly came in a form of a veto: seven municipalities organized local referenda between 2002 and 2004, and the number of 'no' votes prevailed all the time.

The project originally was brought together by nine municipalities, but soon after that many other villages and towns joint to the consortium, finally 59 municipalities took part in the program. They planned to set up a new regional landfill that would handle the municipal solid waste of the region in Püspökszilágy. In this village waste management facilities were not unknown to the inhabitants; a low-level nuclear waste repository was already in operation there. In spite of this (or may be just because of this) the people of Püspökszilágy rejected the siting of a landfill in 2002. This was the first rejection that was followed by seven others; only one village – Valkó – would have supported the new site, but in this case the Ministry for Environment and Water vetoed the construction. Numerous stakeholder groups were involved in the siting process; inhabitants were supported by local and national environmental activist groups. The referenda showed that most people refused to have a waste management facility even with valuable economic compensations. Some critics claimed that rival waste management companies also enhanced the social conflicts since they opposed the new facility to be built from a business point of view (Kiss, 2005).

At the beginning of 2004 decision makers seeing the series of failures declared that there is no need for a large regional landfill, but modernization and expansion of existing facilities are sufficient for handling waste management problems of the county.

4.2 The landfill of the North-Balaton region

The North-Balaton waste management project similarly to the formerly introduced North-east Pest County program had a long story in the first years of the new millennium. Veszprém with its 64,000 inhabitants plays a central role in the North-Balaton region; it is a cultural and economic centre. The increasing level of the municipal solid waste produced in the city worried city officials and they set up alternative courses of action in order to solve the problem. Their first idea was to expand the existing landfill of the city, however due to environmental reasons (the expanded landfill would have been built on a karstic area where no waste management facility can be placed) they had to give up this plan.

In Királyszentistván, which is located 10 km away from Veszprém, in 2001 a local referendum was held where the people of the tiny village (it has approximately 500 inhabitants) rejected the idea of a new regional landfill. The village seemed to be completely divided: the difference was so small that only ten votes decided. Other municipalities such as Ajka or Nagyvazsony earlier expressed that they would not welcome a noxious facility. However, the biggest disputes occurred in Szentgál where the people supported the siting of a new landfill, yet the neighbouring towns and villages heavily opposed the plans. Although the proposed facility would have been located officially in Szentgál, it would have been closer to the houses of the neighbouring municipalities; hence the negative consequences such as smell, environmental risks, increased traffic, etc. would have affected mostly them. Moreover, the project management team offered a compensation only for the locals of Szentgál (among others they offered a large sum of money for the renovation of the local elementary school), but the neighbours were neglected in this process. Although the people of Szentgál voted with yes in a referendum in 2003, the facility was never built there

since the surrounding villages sued the investor company. By launching a litigation case the neighbouring villages created a dead end for the investment in Szentgál, since the construction could not have been started till the court did not make a decision. However, time pressure was enormous on the investors: they had to start building the site till 2006; otherwise the EU funding would have been lost.
Surprisingly enough the project management team after the fiasco, went back to Királyszentistván where the initiative was also a failure several years before. Nonetheless they implemented a more efficient strategy this time, they informed the people, and offered compensation for some neighbouring villages as well. During the second referendum the project was now supported, and the landfill was constructed, and finally opened in 2010.

5. Social conflicts around hazardous waste siting

As Table 1. highlighted the amount of hazardous waste has been decreasing recently; in 2008 it dropped bellow one million tons (Hungarian Central Statistical Office [HCSO], 2010). In this section two emblematic case studies will be introduced: (1) the brief history of the battery waste recycling facilities, and (2) the DDC Cement factory case where hazardous waste was utilized as an alternative fuel in the factory.

5.1 The brief history of the battery waste recycling facilities

The case of battery waste recycling facilities was discussed in several works, although each case study focused on a different stage of the events (Szirmai, 1999; Szántó 2010). This case, which has been dragging on for years, is the model example of Hungarian siting decisions, which exemplifies almost every one of the errors which can be committed by the decision-makers, while also shedding light on the institutional, political and social factors influencing the siting of undesired facilities. A waste battery recycler is a typical NIMBY facility: the majority of the Hungarian society accepts its necessity in general terms (not to mention the international disapproval of shipping hazardous waste across the borders), but will show fierce local opposition to any specific siting attempt. Without a recycling facility the waste batteries must be exported to the neighbouring countries such as Austria and Slovenia and waste battery containing a valuable amount of lead are reused there.
In Gyöngyösoroszi, most objections concerned the prospective technology, beside the already high environmental load of the area; in Komló, the plan was condemned to failure by the counter-reactions of the adjacent settlements, which thought that they would share the burdens with the residents of Komló, whereas the benefits (local taxes, jobs) would go exclusively to the host settlement.
In Monok, a settlement at the gate of the Tokaj vine region, siting efforts failed because the local viticulturists felt that they threatened the reputation of the Tokaj wines, and did everything to kill the siting by their protest actions. Nevertheless, the inhabitants of Monok led by their mayor supported the idea to have a hazardous facility nearby since the new investment would have brought new jobs (around 200 employees would have been hired if the plant had been constructed) and growing tax revenues for the village. The unemployment rate in Monok exceeds the national figures significantly and the incomes of the local people just lag behind the one of more developed regions. Despite the differences in the underlying reasons, the success of the opposition was due in every case to forming a local coalition and to pooling the local interests (Szirmai, 1999) and, with the exception of Monok, the cases concerned confirmed again that investors like to site facilities at locations which have already hosted (voluntarily or under some constraint) a hazardous facility of

some sort. A shift in favour of less resistance is, of course, reasonable, but it may be challenged on the ground of the failures and moral problems. Fear from stigmatisation is also discernible in the Monok case: the Tokaj farmers feared – probably with good reason – that their products will be less marketable if it turns out that there is a waste battery recycling facility near the vine-growing region.[2] The investor realizing the hostility of the potential host municipalities had to withdraw. This siting process seems to be a never-ending story since the problem is not solved entirely yet even in 2011.

5.2 Heating with alternative fuel – The case of the DDC cement factory

Contrary to the above-described cases, the Vác cement factory case concerns the introduction of a new technology, not the siting of a new facility. Consequently, it differs in some essential respects from the previous ones. The envisaged introduction of hazardous waste incineration, however, can be conceived of as a special siting decision, and its reception was rather similar to the social conflicts triggered by the prospects of the new facilities of the battery waste recycling plants.

The hazardous waste incineration case of Duna-Dráva Cement (DDC) broke out in 2002. In November 2002, the company announced that the Central Danube Valley Environmental Protection Inspectorate authorised the factory to incinerate waste, including hazardous waste, as part of the cement manufacturing process. The permit applied to an annual 75 thousand tonnes of waste, a substantial part of which could be hazardous waste. The issue of alternative waste incineration had already been raised in the factory previously due to the many foreign experiences demonstrating the applicability and cost-effectiveness of this technology. The announcement was followed by protests on such scale as was unexpected to both the company management and the municipality. Some environmentalist groups disputed the professionalism of the environmental protection examinations, and in November 2002, the Hungarian Green Party started canvassing for signatures and in a short time it collected around 800 signatures from protesters. DDC and Vác municipality organised a forum together with the Vác Environmentalist Society, where it turned out that the factory had been experimenting with the incineration of various acid-resin-containing materials (spent oil, so-called Cemix and Mumix mixtures). (The factory was repeatedly accused of illegal waste-burning, but as a matter of fact they had had a permit for experimental acid resin burning valid until August 2001.) Several appeals were lodged against the resolution of the National Environmental Protection Inspectorate. Some objected to the incineration site being close to a school, and others found it injurious that the waste transports would probably increase the already quite heavy traffic on main road 2.

The Duna-Dráva Cement case culminated in 2003 and 2004. In January 2003, the National Inspectorate cancelled the waste incineration permit with reference to procedural errors, and obliged DDC to have a new impact assessment made. The Inspectorate was of the opinion that public hearings had to be held on cases like that, and the company had to make a full environmental impact assessment. Although the representatives of the company and of Vác municipality repeatedly emphasised that there were no professional arguments against the incineration of waste and in particular hazardous waste, and that the process was in full compliance with the environmental protection requirements, the opposition prevailed. After a

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[2] A local press organ published an article entitled "Chernobyl, too, was believed to be safe" in connection with the siting of the hazardous waste processing plant in Monok (Szántó, 2010). The envisaged investment was often compared to facilities which, although they did not have much in common with waste processing plants, evoked experiences which could stigmatise it.

change in ownership in the nineties, the company tried to break with its previous negative image (the "one of the dirty dozen" nickname), but the image of the smoke-emitting cement factory and the awful dust it produced have never been forgotten by the locals.
In reaction to the protests, first of all an AdHoc Committee was formed to receive the complaints and observations of Vác residents and to forward them to the company management and, vice versa, to relay information obtained on the activity of the company to the population. The members of the six-strong committee included the heads of three Vác-based NGOs, the managing director of the factory, the environmental councillor of the municipality and a citizen of Vác. Although the members were independent, except for the factory managing director, the most prominent opponents were not represented on the committee. A toll-free hot line was installed in the Mayor's Office, where the locals could make announcements concerning the factory. Given the social pressure, the propositions of the civilians were taken into account in the full environmental impact assessment. This process took almost one year. In the meantime, the company came to realise that, to have the new technology accepted, it must open up towards society: they organised open days and pursued more intensive communication concerning the activity of the company and the waste incineration process itself. DDC's operation was shown on the local TV channels, and they, too, introduced a toll-free call number to receive questions and opinions. The company issued a newsletter called *Monitor*, which presented its activity and made public opinion polls to probe the attitude of the population to it. DDC enhanced its already quite impressive sponsoring activity: according to their own statistics, sponsoring expenditures doubled from 2002 to 2004.
The full environmental impact assessment was made public almost one year after the break-out of the events, in September 2003. On 24 February 2004, the Social Control Group was formed, the members of which were recruited mainly from the representatives of the previous AdHoc Committee: its president was the secretary of the Environmentalist Society of Vác (Váci Környezetvédelmi Egyesület), and its 13 members included the managing director of the Vác factory (who used to be on the AdHoc Committee), the representatives of certain civil organisations of Vác, the representatives of the municipalities of Vác and other settlements, and other opinion-leader personalities of the town. The Group was created pursuant to the decision of the mayor and the management of DDC, to ensure comprehensive social control over the cement factory and not in the least to build public trust in the factory. The opposition diminished considerably over the 18 months under study, but it could not be eliminated totally.

6. Nuclear waste and siting

Research in the 1980s revealed that the rejection rate was highest for facilities regarded as definitely hazardous, such as nuclear power plants and incinerators of hazardous waste (i.e. risk factors associated with relatively low probability of occurrence and catastrophic consequences) (Kasperson, 1986; Mitchell–Carson, 1986). In Hungary siting of radioactive waste repositories has not been such a hot issue as it was for example in the United States where in the last decades no nuclear waste management facility has been created. In this section – following the logics of the international literature – the low-level and high-level nuclear waste repositories will be introduced separately since these cases showed different patterns.

6.1 Low level radioactive waste repository

The discussion complications of the siting of radioactive waste from the Paks nuclear power plant fits into the international trends analysing nuclear waste siting, as witnessed by a recent paper (Vári–Ferencz, 2006) which undertakes to summarise the events. Although the authors examine the siting of low-level waste and high-level waste separately, their conclusions apply to both areas.

The case of the Ófalu repository, which became a symbol of the inadequacy of the top-down decision-making mechanism of the socialist regime in the history of low- and intermediate-level nuclear waste siting looking back on a longer past, has made it very clear that the technocratic approach and the consequent total exclusion of the population, a typical feature of environmental decision-making in the seventies and eighties, is untenable (Juhasz et al., 1993; Szíjártó, 1999). Ófalu as a location for the repository was proposed by the Paks Powerplant in 1987. The management of the plant did not inform the inhabitants who protested vehemently against the decision. As the Bős-Nagymaros Dam and the Dorog incinerator case the Ófalu case also a social conflict of the system change at the end of the eighties. Their protest was successful; the power plant had to withdraw.

The case of the selection of the Bátaapáti repository site was a relatively positive example of a new variant of environmental decision-making. That decision-making model was based on screening methods, which first screened the sites which did not conform to the geological and technological criteria, then studied the expected reactions of the population, followed by another screening of the candidate sites on the basis of that survey. This model is worth comparing with the procedure proposed by Swallow et al. (1992). They developed their model in connection with the construction of a solid waste landfill. In Stage 1, the potential sites conforming to certain minimum technical standards are selected; in Stage 2, the candidates are tested against some social requirements. Stage 2 results in a short list of candidates, of which one is selected in Stage 3 through the compilation of a compensation package. The investment site to be selected is the one that will be accepted by the population at the smallest compensation.

In the Bátaapáti case their three-stage model was replaced by a more limited decision-making procedure. In Hungary, the second stage was omitted (in the opinion of Vári & Ferencz (2006), Bátaapáti is obviously not a suitable candidate site for a nuclear waste repository investment due to its agricultural and recreational profile), but the candidate host settlements were highly interested in the problems of the third stage (compensation specification, choice of the host settlement). Vári and Ferencz (2006) note that, after the systemic change the environmental decision-making model shifted quite noticeably from the technocratic to the market model; the investors realised the importance of compensation packages and upgraded their communication, often with the assistance of PR companies. The conflicts frequently turned the suffering stakeholders themselves against one another, and made the candidate settlements compete – due to their vulnerability and economic backlog – for hosting the facilities which in their opinion had detrimental effects (this, on the other hand, is in good agreement with the model of Swallow et al.).

6.2 High level radioactive waste management

Contrary to the previous section, there was no social debate and no definite standpoint was adopted concerning the social factors, in the case of Boda, a candidate for siting high-level nuclear waste. Back in 1986 the Paks Nuclear Power Plant made a contract with Soviet commercial agencies that the Hungarian high-level nuclear waste would be transported to the Soviet Union. Yet, after the Soviet Union collapsed this solution became fairly unstable therefore the power plant started to make research for the creation of a permanent high-level radioactive waste repository (Vári, 2009). The power plant began geological research in Boda in 1995 where the siltstone formation seemed to be suitable for placing high-level radioactive waste into the ground. Although this research was stopped for three years for political reasons in 2002 they were restarted and an underground laboratory was constructed in the following years.

Parallel with the research in Boda, a temporary facility to store spent nuclear fuel was established in Paks. It was another success story from the point of view of the Nuclear Power Plant, as the repository could be constructed near the Plant. The case is discussed in an earlier study by Vári (1996), in which she identifies the following factors as the pledge of success: (1) learning acquired from the fiascos of the Ófalu case, acquisition of high-level political support, (2) efficient PR activity, (3) shaky civil organisations on the opposite side, (4) material interest of the residents of Paks in the operation and even expansion of the power plant and (5) the population got accustomed to the proximity of the plant.

Boda is a small village with less than 500 inhabitants in a mountain region. In 1996 an association was formed with other five surrounding municipalities. This association socially controlled the research project, continuously informed the people and made decisions about the funding that was received from the Paks power plant (Vári, 2009). The communication had been intensive from the beginnings, a professional communication agency was assigned to create and implement an effective communication campaign. They regularly issued newspapers about the topic and they built an Information Office and Park in Boda. Interestingly, the residents of Boda seem to have become immune to the series of new communication efforts, and the issue of the high level radioactive waste repository has become less and less interesting to the public.

7. Discussion

Anna Vári, one of the most renowned Hungarian representatives of research programmes based on case studies devoted a decisive segment of her research activity to environmental conflicts related to siting and to the investigation of the related risk perceptions. Her works escorted the emblematic Hungarian environmental conflict cases, so to say: the scandals of the construction of the M0 ring road around Budapest; the severe controversies of the cyanide pollution conflicts in 2000, the open questions of the Ajka Power Plant investment and the problems concerning the siting of nuclear waste originating from Paks. Rather than providing a mere case description, the publications of Vári and her colleagues investigate the case in its broader context and draw more general conclusions from its progress. Vári identifies several reasons of the fiascos. Firstly, the objectives and the general plans have not been made clear. Secondly, in several cases, lack of alternatives put things on a forced track which was not acceptable for the stakeholders. It was observed on several occasions that the public was excluded, that no trust existed, and this has often generated tension even among the opponent civil organisations. The predominance of the technocratic approach (almighty planning staff), and the inadequate handling of the compensation packages (e.g. to "buy" the municipalities) has but intensified the opposition. Although foreign experts and PR firms were hired to help with the siting decisions and to communicate them, their success was at least dubious; in Vári's opinion, the foreign instruments and results have not been adapted to the Hungarian circumstances. These conclusions seem to apply not only to the motorway construction projects but, in a broader sense, to Hungarian siting decisions related to waste management facilities as well.

Risk communication has been a priority issue in Hungary, but there are different approaches as to the best manner of this communication. After transition, the new governments wanted to promote authentic information provision on siting matters by establishing the necessary legislative background, but it is often quite noticeable that this legislation is not enforced. Investors often neglect to inform local inhabitants about the potential drawbacks of the planned waste management facilities, and local people are seldom invited into the decision making process. The most serious deficiency of the communication policies was the predominance of one-sided communication. Despite every effort to develop bilateral communication and to integrate some external opinions (for example through a monitoring group in the Vác cement case), the companies rarely entered in a real dialogue, and the two sides often missed each other's points. It is not enough to organise fora and discussions: people must believe that the companies will pay heed and listen to their opinion. It takes two to have a dialogue – if one takes the initiative; the other must at least be sufficiently open and receptive.

The fate of the envisaged facility was decided by local referendum in a major part of the siting conflicts. This institution, if coupled with the right to veto, gives considerable power to the local community – in such cases, the host settlement exercises the proprietary rights. The body of representatives of the settlement, on the other hand, is bound only by the so-called binding ("decisive") local referenda, whereas a non-binding referendum will only give orientation to the management of the settlement. Several referenda were held in connection with the siting decisions discussed in the above case studies. A closer analysis of the cases, however, reveals that a negative referendum decision does not necessarily mean that the facility will not be built and, surprisingly, the inverse may also happen: the predominance of "yes" votes does not always lead to the installation of the facility. In Királyszentistván, although the siting of the regional waste repository was rejected twice (once at a public hearing and later on at a referendum), the disputed facility could in the end be built, since the locals voted "yes" on the third occasion. Note, however, that local referenda are often invalid, because less than half of the population goes to the polls. In such cases the decision-making right is usually transferred to the body of representatives.

In Hungary, most line policy issues acquire a very strong political connotation, and environmental issues are no exception. The siting decisions and the related social conflicts often have a political dimension (cf. the siting conflicts concerning the waste battery recycling plants (Szántó, 2010)). Politicians are important actors of social conflicts generated by siting decisions: in Vác for example, the Social Control Group was set up on the initiative of the mayor.

No wonder that the protests against the siting decisions are often espoused by the opposition parties, which obviously expect to gain political advantages thereby. Those in office ever usually support the investors in the hope that job creation and the growing tax and other revenues will strengthen their position in the management of the settlement and bring them extra votes at the subsequent elections. The central administration is usually also interested in the realisation of the siting decisions because, as in the case of the local managers, a successful investment may generate political capital for them. It would be exaggerated and an oversimplification to say that the siting cases are political games pure and simple, aimed at the acquisition of political power, but the attitude and behaviour of the government and the opposition politicians, respectively, in relation to the cement factories clearly diverge in line with their respective political orientations. Both parties must take this – the forceful intervention of politics in the siting conflicts – into account, as well as the fact that some will try to exploit the possibilities inherent in the roles played by the politicians.

The entry of party politics in the siting cases is, of course, not a surprising phenomenon, and certainly not a specific Hungarian feature. It is, however, worth separating the party political skirmishes from the political processes in the broader sense which shape the life of a settlement. The appearance of a new facility in a settlement may be a decisive affair, so it is quite understandable that it is part of the public discourse there and the various interest groups express their positions concerning the issue. It is a problem, on the other hand, if politics dominates these cases and drives the disputes into a party policy channel, because that makes it impossible to develop an open dialogue between the actors.

8. Conclusion

This chapter dealt with the development of the social conflicts around waste management facility siting in the last two decades of Hungary through case study research. It analyzed the most important cases of the last twenty years and explored what were the causes and implications of these social conflicts. Research methodologies based on case studies are especially popular in the area of siting decisions laden with environmental conflicts, not only in Hungary, but also in many other parts of the world where such research is carried out. This is no accident. In these cases there are many viewpoints, and many actors with different motives, which often provoke serious social conflicts.

The above review of the most important Hungarian case studies warrants the conclusion that the main roots of the research of siting conflicts are sociological ones. In Hungary, the sociological approach seems to be the most relevant of all the main trends manifesting themselves in the international technical literature. Almost every researcher states the domestic siting decisions cannot be discussed without speaking of the role of the social and political impacts. Let us risk the statement that this is a typical Central and Eastern European, rather than specifically Hungarian, phenomenon. The fact that transition and the surge in environmental conflicts occurred at the same historical moment anticipated the lead role of the sociological approach beside the psychological and economic ones in the analyses in these countries. The cases of the past years, on the other hand, highlight that community information programmes, especially the ones deploying PR means, sometimes manage to convince the locals that the given facility implies no special hazards to them. These techniques, however, cannot replace the participative methodologies recommended to date by most Western European and American researchers of the topic (Vári, 1997).

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4

Planning the Management of Municipal Solid Waste: The Case of Region "Puglia (Apulia)" in Italy

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1. Introduction

The management of wastes, in particular of municipal solid waste, in an economically and environmentally acceptable manner is one of the most critical issues facing modern society, mainly due to the increased difficulties in properly locating disposal works and complying with even more stringent environmental quality requirements imposed by legislation.
In addition, in recent years the need to achieve sustainable strategies has become of greater concern, also because some traditional disposal options, like landfilling, are progressively restricted, and in some cases banned, by legislation, so the development of innovative systems to maximize recovery of useful materials and/or energy in a sustainable way has become necessary.
The sustainability concept is today widely used when speaking of the development of human activities, especially in the environmental field. According to the original definition of 1987 by the United Nations, which defined sustainable developments as those that "*meet present needs without compromising the ability of future generations to meet their needs*", sustainability occurs when natural or renewable resources are consumed less than, or at least equal to, nature's ability to replenish them.
As shown in Figure 1, for achieving effective sustainability, three elements of fundamental importance are strongly interconnected to one another and cannot be separately considered (Adams, 2006). They are:

- the environmental element (environmentally robust, and supported by consistent and applicable normative and legal requirements),
- the economic element (economically affordable, technologically feasible, operationally viable),
- the social element (socially desirable, culturally acceptable, psychologically nurturing).

That's particularly true for Western European countries, due to land scarcity and population density, but will also become in short time of major concern for Central and Eastern European Countries that already joined the European Union (EU), or are preparing to do that.
From a general point of view, the waste management policy should be addressed to both the development of management procedures able to reduce the waste mass production, and the application of reuse options instead of simple disposal ones.

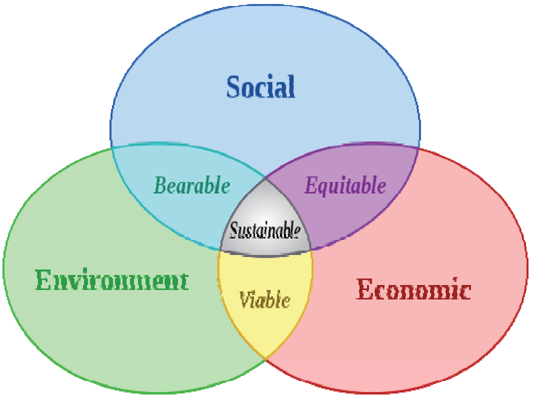


Fig. 1. Interconnection of elements affecting sustainability (Adams, 2006)

In this chapter the criteria staying behind the definition of plans for municipal solid waste management in the Region Puglia (Apulia), south-east of Italy, are discussed, together with the treatment and disposal facilities of the first installation fulfilling the requirements of the above mentioned plan.

2. Technical aspects

Main options for the management of municipal solid waste (from now on abbreviated as MSW) include stabilisation, incineration and landfilling (Williams, 2005).
Stabilisation can occur in aerobic environment (stabilisation, composting) or in anaerobic one (digestion).
Stabilisation and composting involve the aerobic degradation of biodegradable organic waste which allows to obtain a stabilised product to be utilised, after maturation or curing, for agricultural purposes. However, to obtain a compost of good quality, the organic (wet) fraction of the waste requires separation from the other (dry) fraction; if the organic fraction is not separately collected, but separation takes place in mechanical separation plants, then the process can be used to obtain a stabilised wet fraction which can be usefully reused for environmental restoration purposes or as a cover material in landfill operations.
Composting has the advantage of producing a safe and hygienic product which can be easily stored, transported and used on times and sites different from those of production. Therefore, the separate collection of the MSW wet fraction is a fundamental prerequisite for a successful composting operation (Spinosa, 2007a).
The main operating variables affecting the process performance are the moisture content (optimal 50-60%) and the carbon to nitrogen ratio (optimal 25-30), to avoid slow composting (at high C/N ratios) or ammonia volatilization (at low C/N ratios).
Anaerobic digestion takes place in a closed reactor in the absence of free oxygen with the production of gas (biogas), rich in methane, and of a solid residue that can be used for agricultural purposes. The conditions are similar to those occurring in a landfill, but better controlled. Also in this case, the separation of the wet fraction from the dry one is required for good performances.
Incineration and other thermal processes, require that economics be carefully evaluated, but could be a cost-effective solution in large urban areas, where the distance to landfill site

makes transportation prohibitively expensive, and when restrictions on landfilling are imposed. Further, thermal processes can usefully deal with materials which do not meet beneficial use requirements (Spinosa, 2007b).

Potential advantages of high temperature processes include reduction of volume and weight of waste, destruction of toxic organic compounds, and potential recovery of energy.

Most important physical-chemical characteristics affecting thermal processes performance are the dry matter and the volatile solids contents. Dry matter affects both fuel requirement and exhaust gas production, while the volatile solids content is important because it affects the calorific value, i.e. the monetary value of the material.

For above reasons, screening and stabilisation of unsorted MSW is a valid pre-treatment process to obtain a combustible dry fraction with good calorific value.

An aspect that needs to be put in evidence regards the sequence of waste separation and stabilisation processes. Separating the waste materials before stabilisation involves smaller area or cell volumes because only the wet fraction is to be stabilised, but organic pollution of the dry fraction could remain unacceptable for its subsequent handling. On the contrary, a better separation is obtainable after stabilisation of the unsorted waste: higher treatment volumes should be in this case required, but the overall performance of the treatment system results generally higher.

Landfilling of municipal solid wastes is a well known and consolidated practice; it is a convenient solution where enough space is locally available at reasonable disposal fees. A landfill is also the necessary support to all others waste handling systems for the final disposal of materials no more eligible to reutilization and during shutdown periods for maintenance and/or emergency (Spinosa, 2005).

However, organic matter deposited in a landfill is not available for agricultural needs, but the production of landfill gas (biogas) is allowed. Biogas, if not captured, considerably contributes to the greenhouse effect, because it is mainly composed of methane, which is about 20 times more powerful than carbon dioxide in terms of climate change effects. Therefore, wastes must be subjected to treatment before landfilling, where treatment means the physical, thermal, chemical or biological processes, including sorting, that change the characteristics of the waste in order to reduce its volume or possible hazardous nature, facilitate its handling and enhance recovery. In particular, landfill gas should be treated and used to produce energy, other ways it must be flared.

Another aspect to be considered is that MSW is often handled by following routes different from those of other organic wastes, mainly sewage sludge, each waste under the responsibility of different authorities, with the consequence that specific technical problems and considerable diseconomies arise. For this reason, integrated co-management systems for waste handling, including such operations as composting, incineration and landfilling, should be developed to allow optimisation of operating modalities and reduction of costs to be obtained (Spinosa, 2008).

Composting is the typical process in which the different characteristics of solid wastes and sewage sludge can be usefully integrated to obtain a final product of better quality, because the relatively high solids content and the carbon to nitrogen ratio (C/N) of solid wastes can counterbalance the low solids concentration and C/N ratio of sludge.

In co-incineration, sewage sludge drying can take place at expenses of the excess heat recovered from solid waste combustion, but greater attention in designing and operating furnaces and exhaust gas abatement systems is required.

Finally, co-landfilling, if permitted by national regulations, allows a faster stabilisation, a better leachate quality and a higher biogas production to be obtained, but the operating modalities must be carefully planned due to scarce physical consistency of sludge.

3. Regulatory aspects

Italy is member of European Union since its establishment in 1957, so national legislation must be issued in agreement and/or application of European Directives.

3.1 European

From the European Union (EU, Figure 2a) regulatory point of view, the Directive 91/156 on wastes, also designated as the "Waste Basis Directive", has been of outstanding significance, as it is always to be observed even with the application of any other specific regulations. This means that the particular requirements deriving from other Directives addressed to particular waste groups additionally apply to general regulations deriving from above Directive. In particular, liquid and solid wastes, and sludge must fulfil the requirements imposed by specific normative, such as the:

- Directive 91/271/EEC on the treatment of urban wastewaters;
- Directive 86/278/EEC on sludge utilization in agriculture;
- Organic Farming Regulation 91/2092/EEC;
- Landfill Directive 1999/31/EC;
- Commission Decision 2001/688/EC for the ecolabel for soil improvers and growing media;
- normative on incineration of waste (e.g. Directives 89/369, 94/67 and 00/76), when applicable.

Further, to favour the material and energy recovery options instead of the simply disposal ones, the Directive 99/31 introduced targets for the reduction of biodegradable municipal waste to be landfilled as follows: reduction by 2006 to 75% of total biodegradable municipal waste produced in 1995, reduction by 2009 to 50%, and reduction by 2016 to 35%.

However, above legislation has been primarily addressed to generally reduce the impact of waste on the environment, while a limited attention has been given to the positive aspects of biodegradable wastes (Marmo, 2002).

Within this framework, a crescent interest has been focused in recent years towards the recovery aspects of biodegradable wastes, so that the development of a Biowaste Directive has been undertaken aiming at promoting the biological treatment of wastes by harmonising the national measures concerning their management.

General principles include, among others, the (i) prevention or reduction of biowaste production and its contamination by pollutants, (ii) composting or anaerobic digestion of separately collected biowaste that is not recycled into the original material, (iii) mechanical/biological treatment of biowaste, and (iv) use of biowaste as a source for generating energy.

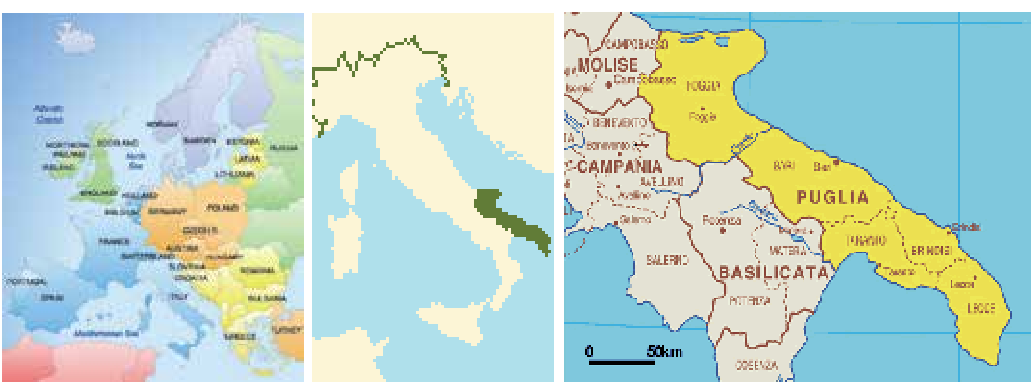
Member States are requested to encourage home and on-site composting whenever there are viable outlets for the resulting compost, and setting up of community composting schemes as a way of involving the general public in the management of their own waste, reducing transport of waste and increasing awareness in waste recycling practices.

However, this process has not yet been completed because subjected to the development of a basis legislation on "soil protection", that will then become the reference regulation for the proposed new legislation on biodegradable waste.

Recently, a new Directive (98/2008/CE) has been issued to regulate recovery activities of wastes, including their energy recovery within the general framework of sustainable integrated management systems.
Further, to properly perform the utilization and disposal operations and correctly fulfil the legal requirements, a fundamental role is played by the definition of standardized procedures for the chemical, biological and physical/mechanical characterization of wastes, and by setting up guidelines of good practice for their management. For this reason, the European Committee for Standardization (CEN), which supports EU Commission in Directives issuing, established, among others, Technical Committees whose scope is the standardization of methods and procedures employed for characterisation of waste (TC292) and sludge (TC308).

3.2 Italian

As told before, each EU Member Country has to develop its local legislation by adopting the communitarian one, but has the possibility to introduce changes specific to local situation, provided they are not in contrast with general requirements of EU normative.
In the case of Italy (Figure 2b), the legislation on wastes is now fundamentally based on the recent Legislative Decree nr. 205, issued on December 2010, that, in application of the European Directive 98/2008/CE, replaced the previous Legislative Decree nr. 152/2006.
With specific reference to MSW, Italian national legislation fundamentally gives the Regions "planning" responsibilities, the Provinces "authorisation and control" ones, and the Municipalities "service operations" duties.
In addition, decisions and/or determinations on specific aspects by other institutional Bodies are necessary to obtain all the necessary permits or authorisations. This fragmentation of competences has brought in some cases to administrative conflict between different public Institutions, so in some Regions a Governmental Commissariat has been established which takes upon himself all responsibilities, by-passing and replacing those of most institutional Bodies.
This is the case of Region Puglia (Apulia), located in south-east of Italy (Figure 2c), where a "Commissariat for waste emergency", then more generally modified in "Commissariat for environmental emergencies", was established since 1997. The position of Commissary has been almost always covered by the President of Region Puglia.



(a) (b) (c)

Fig. 2. Maps of EU (a), Italy (b), and Region Puglia (Apulia) (c)

4. The solid waste management in Puglia

With the Commissary Decree nr. 296/2002, based on the Italian Legislative Decree 22/1997 that was at that time a comprehensive law regulating waste issued in compliance with various EU directives, the "Regional Plan for Solid Waste Management" was approved, and then completed and adjourned by the Commissary Decree 187/2005. In previous Decrees, issued on 1997 and 1998, Guidelines for the mechanical-biological treatment of residual waste remaining after separation at source of selected fractions were introduced.
The Region Puglia has an extension of about 20,000 km^2 and a population of a little bit more than 4 millions. The Region is characterised by a coast length of about 800 km, and a hill-shaped profile, also including both vast flat areas and mountains (up to 1000 m) ones.
Administratively speaking, at the time of Plan issuing, the Region included 5 Provinces (i.e. Bari - the Region's Capital, Foggia, Brindisi, Lecce, and Taranto). A new Province, the BAT (Barletta, Andria, and Trani) Province, has been established in 2004 and really started to operate in 2009. This Province, which includes 10 towns originally part of northern Bari Province and southern Foggia Province was obviously not included in 2002 planning, but will be considered in the revised waste management Plan to be approved in 2011.
Basically, above mentioned Decrees (296/2002 and 187/2005) require the:

a. development of "source separation" schemes with the target for 2010 of 55% of MSW separately collected to be handled for material recovery,
b. "biostabilization" of urban waste, remaining after source collection, followed by separation of a treated wet fraction to be landfilled (abbreviated in RBD) or used for environmental purposes (RBM), and of a dry fraction (FSC) to be used for the production of refuse derived fuel (RDF).

Regarding the biostabilisation treatment, the following two options can be adopted:

4.1 Option 1

This option includes the following operations:

- Pretreatments, e.g. storage, moderate shredding by systems compatible with the characteristics of organic materials, ferrous materials separation;
- Biostabilisation for an approx period of 2-4 weeks, depending on the technology adopted, to obtain a material having a Dynamic Respirometric Index (DRI) of max 800 mg-O_2/kg-VS*h;
- Selection/Screening, at max 80 mm;
- Landfilling of the undersized fraction (RBD), at an amount not higher than 35% of the untreated urban waste;
- Processing of the oversized fraction (FSC), amounting to about 40% of the untreated urban waste, to produce refuse derived fuel (RDF).

4.2 Option 2

This option includes the following operations:

- Pretreatments, e.g. storage, moderate shredding by systems compatible with the characteristics of organic materials, ferrous materials separation;
- Biostabilisation for an approx period of 2-4 weeks, depending on the technology adopted, anyway to obtain a material having a Dynamic Respirometric Index (DRI) of max 800 mg-O_2/kg-VS*h;
- 1st Selection/Screening, at max 80 mm;

- Maturation/Curing of the undersized fraction for an approx period of 4-8 weeks, depending on the technology adopted, to obtain a material with a DRI of max 400 mg-O_2/kg-VS*h;
- 2nd Selection/Screening, at max 25 mm;
- Utilisation/Recovery of the undersized fraction, at an amount of about 25% of the untreated urban waste, for use as landfill cover material or land reclamation (closed mines, etc.);
- Processing of the 1st and 2nd oversized fractions (FSC), at an amount of about 45% of the untreated urban waste, to produce RDF.

The overall bloc diagram of such integrated system for management of unsorted MSW is shown in Figure 3. As told, all MSW is biostabilised before selection/screening to get a more efficient separation and reduction of possible malodours.

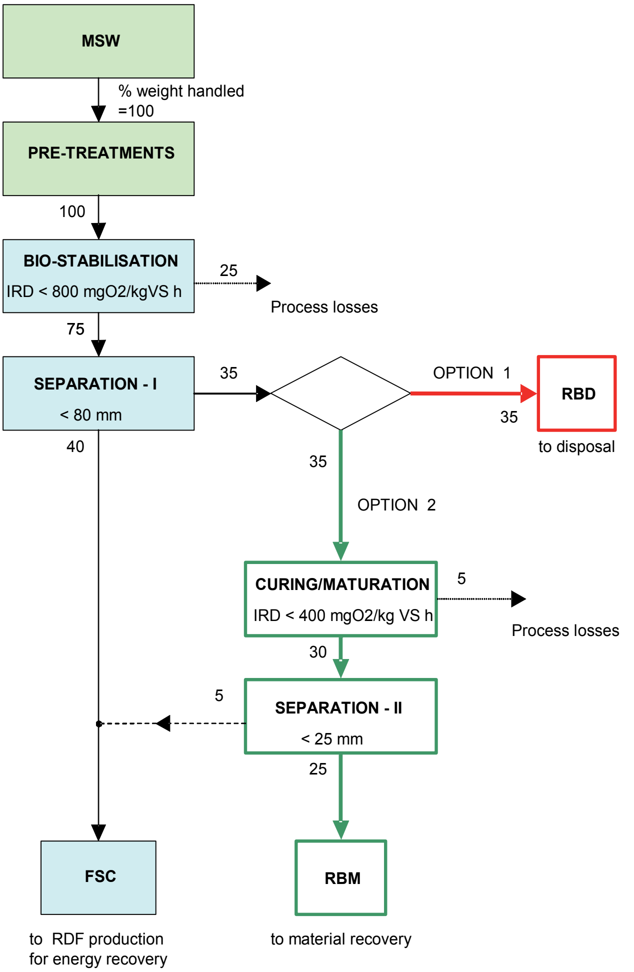


Fig. 3. Bloc diagram of integrated system for management of MSW

For the practical application of above schemes, the regional territory has been divided in 15 "Optimal Territorial Basins" (OTB): 4 in Province of Foggia (FG/1, FG/2, FG/4 and FG/5), 4 in Province of Bari (BA/1, BA/2, BA/4 and BA/5), 2 in Province of Brindisi (BR/1 and BR/2), 2 in Province of Taranto (TA/1 and TA/3) and 3 in Province of Lecce (LE/1, LE/2 and LE/3). Each OTB is served by treatment plants for:

a. "qualification" of recyclable fractions deriving from "source separation or separate collection" of MSW;
b. "pre-treatment" of residual waste deriving from conventional "not-separate collection";
c. "biostabilisation" of above pretreated waste, followed by "mechanical separation" into a "wet fraction" and a "dry fraction", being the former (RBD) landfilled or submitted to further curing for the production of RBM to be possibly reused for environmental purposes, the latter (FSC) processed for conversion into RDF;
d. "landfilling" of process rejects or untreated waste during shutdown periods for maintenance or emergency.

Operation of above point a) has the purpose to have a higher amount of selected fractions of better quality just to give them a higher market value.

It has to be observed that, to optimise economic balances, the production of RDF and its utilisation is planned not to be done in all OTBs, but in a few centralised Centres serving more OTBs. This is the case of Province of Foggia, where 1 RDF production Centre is planned to serve 4 OTBs, of Province of Brindisi to serve 2 OTBs, of Province of Lecce to serve 3 OTBs, of Province of Taranto to serve 2 OTBs, and of OTB BA/1 serving also OTB BA/4.

At the time of writing 10 treatment plants are in operation (OTBs of FG/3, FG/4 and FG/5; BA/2 and BA/5; TA/1 and TA/3; LE/1, LE/2 and LE/3) and 1 is completed and ready to start (OTB of BR/1).

4.3 Guidelines

To guarantee uniform technical designing of plants in the different OTBs, specific Guidelines for each treatment section have been issued by the Commissariat Offices (Commissariat for waste emergency, 1997, 1998a, 1998b, 1998c).

Guidelines require that, besides main working structures, all Centres shall be provided with facilities destined to Support Services, subdivided into Management Services and Technical Services.

The Management Services include:

- weighing;
- waste classification and recording;
- guardhouse;
- administration;
- social services for personnel,

while the following services and/or technological installations belong to the group of Technical Services:

- motive/driving power and lighting electric installations;
- water supply system for drinking, hygienic and services uses;
- effluents treatment plant;
- surface water disposal system;
- fire protection system;
- earth plant and lightning strokes protection systems;

- storage, handling and materials loading/unloading areas, with sizes and characteristics suitable for passage and operation of lorries, trucks and trailers;
- parking areas for vehicles and demountable containers, spare parts store.

4.3.1 Centres for qualification of recyclable fractions from separate collection

Such Centres shall be used for paper and cardboard, plastics, glass, aluminum cans, ferrous and non ferrous metals (Commissariat for waste management, 1997).

The main equipment is the selection system, essentially consisting in a belt conveyor located on a platform equipped with a sound-proof cabin and an air-change system. Operators, standing at belt side(s), manually pick up the different fractions and store them in containers placed below the belt. From the material remaining after the above selection, the ferrous material is separated by a permanent magnet deferrization system, whilst aluminum and non ferrous materials by an eddy current separator. The other materials deriving from the selection which cannot be recycled are discharged in special containers, compatible with the material itself, for disposal at authorized plants. Paper, cardboard and plastics must be pressed and pressing devices must assure, for plastic wastes, their pressing in bales sizing 120x80x80 cm, each weighing 100-140 kg. A baling press for the compression of aluminum cans must be also installed.

As far as the storage sites of glass, plastics, paper, cardboard and cans are concerned, Guidelines require the realization of 3 sides walls cells in reinforced concrete with a height of 2.5 m, width and length not lower than 3 m and 6 m, respectively, smooth concrete pavement and protection against wear and tear, with a light slope (max 2%) towards the open loading side, with a grating for collection and conveying of meteoric waters. The storage sites for processed plastics and paper/cardboard must have a capacity sufficient for the storage of, at least, a quantity corresponding to 2 units of useful load, equivalent to 200 bales, while the storage capacity of processed cans must be sufficient for the storage of at least a quantity correspondent to 1 useful load, equivalent to 30 tons.

The Centres must be also equipped with a 80 t weighing balance with 18x3 m^2 platform, and with additional equipment for materials handling, loading/unloading, storing, etc., in number according to the potentiality of the Centre.

4.3.2 Centres for selection of unsorted wastes

Such Centres allow waste residuals from separate or undifferentiated collection or from separate dry/wet collection to be delivered (Commissariat for waste management, 1998b).

Such plants must be located at least 1,500 m far from the limit of urban agglomerations and of important or touristic areas and at 2,000 m far from hospitals, health or thermal centres. Providing that all sectors must be equipped with suitable systems for odors and dust control, in case using biofiltration apparatus, collection and storage of entering waste to be sent to selection must occur in a confined space. The size of such sectors must allow the storage of the maximum quantity of daily production for a period of 3 days, at least.

The separation system of the wet fraction from the dry one must allow (i) the bags breaking and the waste size reduction preferably through shredding systems, excluding thin comminuting techniques, incompatible with the organic materials nature, (ii) the separation, through screening, of the wet fraction (undersize) from the dry one (oversize), (iii) the separation of ferrous and non ferrous metallic materials.

Above system must be located in a shed with an industrial type pavement, water-proof and suitable for the passage of mechanical means, as well as with a wastewater collection and disposal system.

Residuals from separation must be stored in special containers or tanks or piles properly protected, compatible with the material characteristics for their subsequent treatment or disposal at authorized plants. The size of the storing sector must allow a storing capacity of the separates combustible material corresponding at least to 7 days, or in such a way as to avoid any risk of hygienic and sanitary problems.

4.3.3 Centres for stabilisation / composting.

Such Centres allow solid waste residuals from separate collection and/or of separated organics to be stabilised. As told, good quality compost can be obtained only if the organic fractions are separately collected.

Such plants must be located at least at 2,000 m far from the limit of urban agglomerations and of important or touristic centres and at 2,500 m far from hospitals, health or thermal centres. All sectors must be equipped with suitable systems for odors and dust control, eventually using biofiltration apparatus, while the collection and storage of entering waste to be sent to selection must take place in a confined space. The size of such sectors must allow the storage of the maximum quantity of daily production for a period of at least 3 days (Commissariat for waste management, 1998a).

Preliminary treatments shall allow the (i) size reduction of input waste, using systems compatible with the organic materials nature, (ii) selection of ferrous and non ferrous metallic materials, and (iii) e separation, through screening, of the other non processable fractions (oversize).

The working cycle includes the two phases of primary biooxidation and curing, which must take place in aerated windrows or closed reactors or mechanized vessels or confined piles. Reactors and vessels must be tight, and the surfaces which the piles are placed on must be water-proof and appropriately protected with industrial type floor suitable for the passage of mechanic means. In anycase, wastewater drainage and collection systems, to be sent to water conditioning or to reuse in the treatment cycle are required.

The total duration of the two above processing phases must fulfill normative requirements; in particular, temperature must be kept for at least 3 consecutive days over 55 °C. A sufficient oxygen quantity must be assured to keep the aerobic conditions of the mass through the use of both fixed aeration systems and electromechanical equipments, and handling means and/or mechanical turning machine to turn the material under treatment. A final refining phase is also required to separate the foreign material eventually still present in the mass of treated materials, to make uniform the product particle size and to reach the desired final degree of humidity.

The final product must be stored in containers or tanks or piles adequately protected in order to preserve its quality and agronomic characteristics and to avoid hygienic problems due to recontamination. Packaging in bags with label in compliance with the law is recommended.

4.3.4 Centres for production of refuse derived fuel

Centres for production of RDF are plants which get the selected fractions of fuel material (e.g. FSC) for their transformation into a solid product to be reused for energy purposes in existing industrial plants or in dedicated ones (Commissariat for waste management, 1998c).

In this case too, all sectors must be equipped with suitable systems for odors and dust control, eventually through biofiltration apparatus. The collection and the storage of

materials to be sent to RDF production must take place in a confined space, dimensioned to allow the storage of the maximum quantity of daily production for a period of at least 7 days. The flooring of the shed must be of industrial type and equipped with a washing water and wastewater collection and disposal systems, in conformity with the applicable regulations.
The production of RDF, to be realized in a suitable closed shed, must allow the (i) separation of the dry fraction into light, thin and heavy fractions (ballistic systems or equivalent ones), and (ii) production of a material in compliance with the quality standards established in the agreements with the users (densifying systems or equivalent ones).
The final product must be stored in containers or vessels or piles adequately protected and with a volume suitable to the Centre potentiality; in anycase it must assure a storage capacity corresponding at least to 7 days of production.

4.3.5 Centres for energetical utilisation of refuse derived fuel

Centres for energetic utilization of (RDF) are plants which receive the selected fractions of fuel material separated in the Centres for production of refuse derived fuel for its combustion and energy production. Such plants must be located at least at 1,500 m far from the limit of urban agglomerations and of important or touristic centres and 2,000 m far from hospitals, health or thermal centres.
The characteristics of RDF to be sent to combustion must be in conformity with the current technical standards, including the Standard UNI 9903-1.
All sectors must be planned in order to reduce dust, volatile organic compounds and odors emissions, according to the best technologies available. The collection and the storage of materials to be sent to combustion must take place in a confined space, dimensioned in order to allow the storage of the maximum quantity of daily production for a period of at least 7 days; the plant must be equipped with specific devices for the abatement of particulate/dust, NO_x, HCl, HF, SO_2, organic micropollutants, and other inorganic pollutants.
The other technical requirements are:

- stack height able to assure a good dispersion of pollutants and to protect human health and environment;
- pavement and floorings of industrial type, equipped with washing water and wastewaters collection systems;
- suitable energetic recovery section under thermal or electric form, with total efficiency not lower than 20% with regard to electric energy production, to be calculated according to the real value of RDF lower calorific value;
- measurement and recording of main working parameters of the energy production plant;
- ash and slag storage in containers or vessels or piles adequately protected and with a volume able to assure a storage capacity corresponding at least to 7 days of production;
- quantification and characterization of mass flows coming out from the Centre;
- data visualization system to the public.

For handling the materials treated in the Centre, the same equipments of other above mentioned Centres must be available.

5. The Massafra plant

The first plant complying with requirements of the Puglia waste management regional plan was that located in Massafra, serving the OTB TA/1 (Photo 1). The plant, whose technical specifications are summarised in Table 1, was built in 2003 and operated since 2004 by CISA s.p.a., so has now cumulated almost 7 years of successful operations.

Photo 1. General view of the Massafra plant

Authorised capacity	110,000 t/y
Operating days	312 d/y
Daily capacity	350 t/d
Operating hours of mechanical systems	12 h/d
Throughput capacity	30 t/h

Table 1. Technical specifications of Massafra plant

Typical composition of RSU treated in the plant is shown in the following Table 2.
Main constituents of the plant are:

- waste receiving area with weigh-bridge;
- two-floors building for waste receiving and production of RDF, being the section for waste receiving elevated of 2.5 m with respect to that for RDF production;
- two-floor building for offices and general services with controlling, monitoring and supervision systems located on the second floor;
- building for biostabilisation of waste separated from that for production of RDF by a 10 m width road; this building includes a total of 13 biotunnels, being 4 of them possibly utilized for RBM or compost production, and annexed auxiliary equipments, storage containers/boxes for materials to be stabilized, and feeding system for wet-dry separation and production of RDF;
- biofilter located close to the building for waste receiving and production of RDF, but at the opposite side of the offices.

All the external access areas and the operating ways and roads are fully paved, and all the plant area is confined by walling and wire fence.
All the produced RDF is recovered for energy generation at the Appia Energy power station, that is located by the side of the waste treatment plant.

Item	% (according to UNI 9246)
Paper	24.20
Plastics	25.94
Cloth / Fabric	0.76
Wood	1.68
Glass	3.85
Metals	2.07
Inerts	2.66
Organics	10.00
Undersize <20 mm	27.53
Evaporation losses	1.31
Total	**100.00**

Table 2. Typical composition of MSW at Massafra plant

5.1 Biological treatment cycle

The overall biological treatment cycle is shown in Figure 4.
Receiving area
The MSW conferring occurs in a closed building which is maintained under light vacuum; access doors are automatically operated for fast opening and closing. Wastes are downloaded directly from trucks on the pavement of the building, and are handled by a tyred loading shovel; during this operation, the operator of the tyred loading shovel checks the waste to verify the absence of non-processable materials.
Pre-treatment
This operation includes primary shredding and separation of ferrous materials by a 50 t/h slow-speed shredder with hydraulic control. The transferring belt is placed in storage pit, thus making easier the loading operation of materials by the handling means. The transferring speed is regulated by frequency variation.
The shredded waste is then transferred to storage boxes, where is taken by a tyred loading shovel for its loading into the biostabilisation tunnels.
Biostabilisation
The biological stabilization process takes place in 13 tunnels (Photo 2). The process, which includes stabilization and drying, requires 7 to 14 days, depending on the quality of waste. Exhaust air is sent to a centralized biofilter to control odours.
Biotunnels are fully constructed in reinforced concrete, and equipped of an insufflating air system from the pavement, through holes of squared mesh of 40 cm. Air fluxes and process parameters are automatically controlled by a computerized system.
After passing through the material, air is recirculated. Material temperatures are continuously monitored and air fluxes consequently regulated through variation of the cycle of each fan which biotunnel is equipped with. The MSW biostabilisation cycle lasts 7-8 days,

thus allowing a max Dynamic Respirometric Index of 800 mg-O_2/kg-VS*h to be got, useful for subsequent production of RDF.

The phases of the biostabilisation process are:

- hygienisation cycle with temperature continuously higher than 55 °C for at least 3 days; the concrete biotunnels guarantee the uniformity of treatment for all the waste mass thanks to the high insulating index of walls;
- after hygienisation, temperature is maintained at about 50 °C which is the optimal one for the development of microflora and micetes working on organic substance degradation; recirculation of treatment air guarantees uniform conditions of temperature, moisture and aeration of the mass;
- treatment air flow rate is higher than 40 m^3/h per ton of material; this allows availability of enough air for cooling phases so the total time of treatment can be conveniently reduced and time useful for biostabilisation consequently increased.

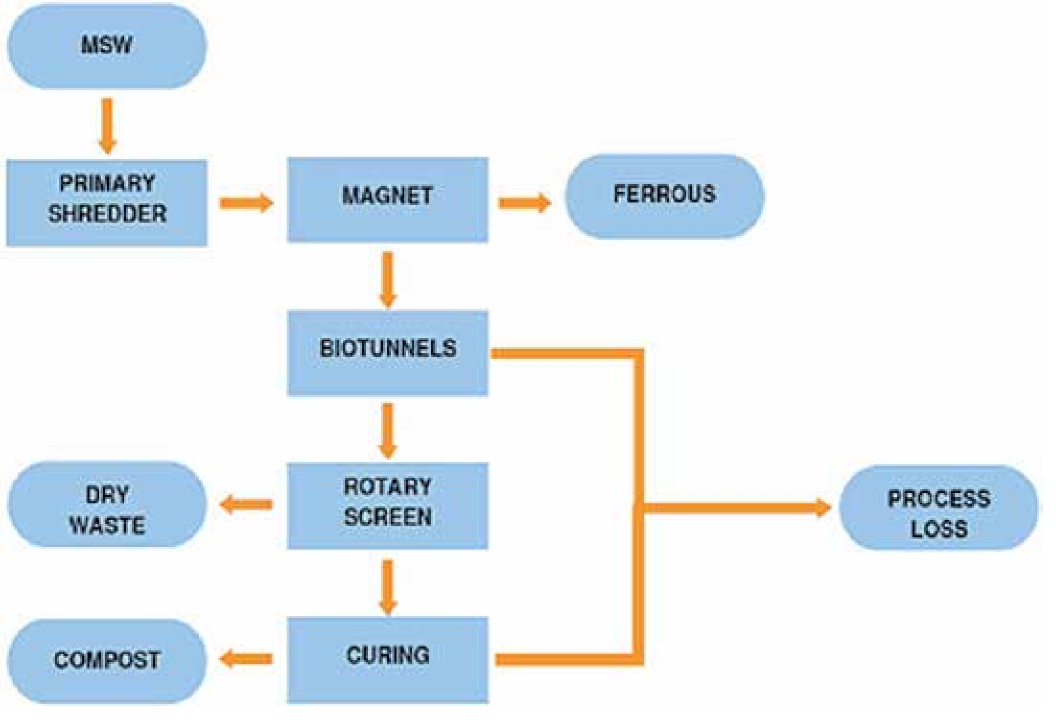


Fig. 4. Bloc diagram of the biological treatment cycle

Parameters controlled in each biotunnel are:

- inflated temperature, directly measured within the pile by thermometric probe inserted through the biotunnel cover;
- temperature of air to be recirculated to the biotunnel and of exhaust air to be treated in the biofilter;
- flow rates of fresh air and exhaust air;
- pressures inside the biotunnel, in air pipes, etc.

At the end of the biostabilisation treatment, the material is transferred to the wet-dry separation section by a tyred loading shovel.

Photo 2. Biostabilisation tunnels

The analysis of control and monitoring system data evidenced that a fundamental requisite for optimizing the biostabilisation process is the material size and homogeneity which strictly depends on the previous shredding operation. Optimal size of the material to be stabilized should range 120–150 mm, thus giving the material the necessary porosity and also guarantee the flaking off of parceled and compressed materials. The type of shredder installed in the plants is able to work in this direction.
In addition, the shredded material has to be submitted to biostabilisation in very short time, just to fully utilize the organic load of waste for a fast and natural temperature increase inside the waste pile during the initial biostabilisation phases. This fact occurs because the fresh shredded material does contain soluble and easy degradable compounds which are utilized by the mesophilic microorganism with production of heat necessary for the subsequent thermophilic phase; a delayed load of biotunnels involves the dispersion of the thermal energy accumulated during the mesophilic phase and, consequently, a not correct development of the process. Such a procedure allows a hygienisation temperature of 55 °C to be reached in 18 h.
For above reason, the choice of a porous pavement in the receiving area, instead of a storage pit, showed to be successful because in a pit the material downloaded from the first trucks remains at the bottom, so it is the last to be treated with possible developments of anaerobic conditions which are dangerous for the process itself, and also causes malodors and leachate release. Aeration through the pavement also avoids the negative effects of pressure on the material, such those caused by systems adopting covered windrow systems.
The determination of Dynamic Respirometric Index on treated material is done on bi-monthly base, while that of raw MSW entering the plant once a year, and any time the collection system is modified or new wastes are conferred to the plant. Sampling procedures are those standardized by the norm 9246 of the Italian standardization body, UNI.
Separation- I
As shown in the process diagram (Figure 3), after biostabilisation the material is screened in a 80 mm openings equipment (Photo 3) where two fractions are separated.

The undersized fraction, or wet fraction, which does mainly contain organic material, is for 80% directly landfilled as RBD, while a 20% portion is cured in an aerated static pile to obtain RBM for subsequent use as cover material for landfill or other environmental purposes.
The oversized fraction, or dry fraction (FSC), is destined to production of RDF.
Curing and Separation-II
The maturation section of the plant, consisting of 4 specific biotunnels, has not been used up to now for the production of compost due to difficulties:

- in supplying the plant of selected organic material deriving from separated collection at source;
- in finding a destination for the compost to be eventually produced, so this section is only used for production of landfill cover material or land reclamation one.

However, above additional biotunnels can be used to expand the overall plant capacity and flexibility.

Photo 3. Selection / Screening equipment

Production of CDR

As told, the oversized fraction from separation is processed to convert it into densified RDF. After ferrous separation, an aeraulic device separates heavy components from light ones, the latter consisting of pieces of plastics, paper, cardboard, polystyrene, insulating material, etc., which are treated by two secondary shredders which reduces the material size thus making it acceptable to be treated by the subsequent horizontal draw bench densifiers, working in parallel, to produce pellets.

A magnetic separator attracts further ferrous material, before the material is processed by the densifiers, and again after them.

Figure 5 shows the bloc diagram of RDF production, and Photo 4 a particular of the pellettizing equipment.

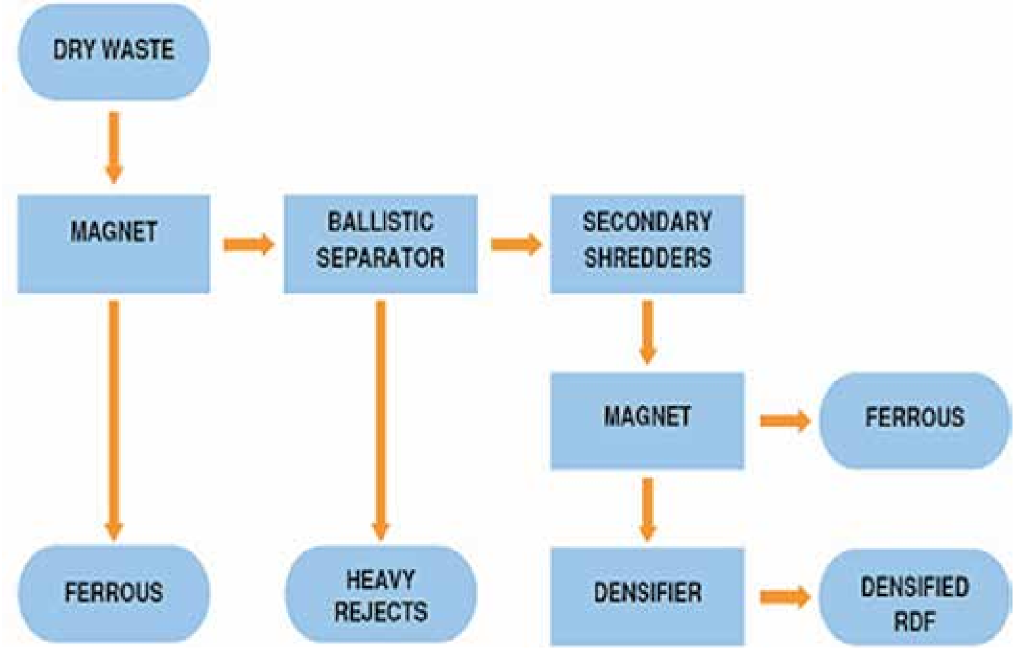


Fig. 5. Bloc diagram of RDF production

The densified material is automatically stored in containers for transporting to the Centre for its energetic utilization, while the heavy components and other manufacturing rejects are belt transferred to storage containers for subsequent disposal at authorized plants

In Table 3 the typical composition of RDF produced by the plant is summarized.

Process control

The control system manages not only all plant devices and equipment, but also records all data of field instrumentations whose analysis made possible the optimization of the entire treatment system.

The plant is also equipped with installations to control dust in the building of production of RDF and air from all plant sections. As a matter of fact, all equipment in the building of RDF production can produce some dust, so they are equipped with suction caps which are connected to a bag filter. The filtered air is then returned to the biostabilisation building which are maintained under light vacuum to avoid air emission outward.

Item	%
Cellulose	20.77
Wood	1.67
Polyethylene LWD	6.27
Polyethylene HD and Polypropylene	3.72
PET	1.91
Polystyrol	1.47
PVC	3.40
Cloth and Fabrics	2.89
Aluminum	0.58
Inerts	0.08
Undersize <20 mm	55.80
Losses	1.44
TOTAL	**100.00**

Table 3. Typical composition of produced RDF

Photo 4. Particular of the pellettizing equipment

All the closed ambients are maintained under vacuum to avoid diffusion of bad odors. Picked up air is utilised in the biotunnels and then sent to the biofiltration system. In the

biofilter plenum, condensate collecting wells connected to the network ending in the corresponding tank of humidification waters for their recirculation are placed.
Leachate from biotunnels, and water drained from all transit areas are collected and transferred to treatment by static grate filter, storage and treatment at authorized plants.

5.2 Energy recovery plant

The energy recovery plant, whose general view is shown in Photo 5, occupies an area of about 90,000 m^2. It is operated by Appia Energy s.r.l.
It consists of the following sections:

- fuel transport and dosing;
- combustion and steam generation;
- combustion gas treatment;
- ash evacuation and storage;
- condensation;
- energy supply and automation.

By means of the pre-heating and superheating phases, produced steam gets pressure and temperature conditions required by the turbine, where it is converted to mechanical energy and then to electric energy through the alternator. All the produced energy is forced into the national energy lines network due to agreement with the network operator.
The low pressure steam from the last turbine expansion stage is condensed to water in air condenser and enters again the thermodynamic cycle.

Photo 5. General view of the power station for energy recovery

Combustion gases, after exchanging their heat with water steam, are submitted to treatment for abatement of polluting compounds.
Steam generator is supported by a steel construction which is covered to protect the generator from atmospheric agents. Maximum height of the structure is 40 m. The stack is 45 m tall and has a diameter of 1.6 m.

The turbo-group is installed in a fire-resistant and sound adsorbing cabin. The interconnecting system to the national electrical network is located near the turbo-group and close to the existing electric lines; it includes a transformer (6.3 - 150 kV).
The following Table 4 summarizes the main operating data of the energy recovery plant.

Produced Energy	Power consumption (auto consumptions)	Energy forced to national network	Gasoil for combustion
kWh	kWh	kWh	Litres
69,672,000	12,524,000	59,040,000	463,286

Table 4. Main operating data of the energy recovery plant

Other power plant data are:
- gross electric power ~12.5 $MW_{e,}$;
- net electric power ~10.0 MW_e;
- thermal power ~49.5 MW_t;
- net efficiency ~21%.

Industrial water needs have been estimated in about 18 m^3/h during the start-up phase and in about 7.2 m^3/h during the operation one, but experience showed that real needs during the operation phase could be as low as 2-3 m^3/h.
The plant can be fed with RDF (main fuel) produced by the MSW treatment plant, and with gasoil (auxiliary fuel) during start-up and emergency periods. RDF consumption is estimated in about 100,000 t/y.
Interferences of the energy recovery plant with environment include gaseous, liquid, solid, noise, and electromagnetic emissions.
Gaseous emissions into the atmosphere are summarized in Table 5. Legal limits are reduced by 20% with respect to the national ones because the plant area is classified at environmental risk due to the presence of many industrial installations.

Item	Units	Value
Wet gases flow rate	Nm^3/h	80,000 – 100,000
Dry gases flow rate	Nm^3/h	60,300 – 89,000
Oxygen (as O_2)	%	~ 11
Exit temperature	°C	~ 170

Table 5. Characteristic emission values of power plant

Reduction of sulphur oxides is obtained within the combustion camera by injection of lime above the fluidised bed. Reduction of nitrogen oxides is obtained through injection of ammonia solution in the post-combustion zone of the furnace. Finally, reduction of acid gases and organic micropollutants is obtained through chemical reactions after dry injection of alkaline substances, such as sodium bicarbonate and activated carbon, in a reaction tower downstream the steam generator. The treatment is completed by a bag filter which retains particulate/dust produced during the combustion process, and residues of the reaction for the abatement of acid gases.

The plant is also equipped with a double system of continuous monitoring of emitted pollutants (CO, NO_2, O_2, Particulate, SO_2, HCI, HF). Other pollutants, such as IPA, Heavy metals, Dioxins, Furans, are also periodically checked.
The authorized limits for stack emissions are reported in the following Table 6.
The system dealing with emissions of liquids is based on appropriate systems which allow most of the liquid wastes to be reutilized in the plant.
Two independent networks respectively collect raining waters and/or those coming from roads, service ways and areas, buildings roofs and coverings, and process waters and sanitary effluents.
Waters from the first network are treated by sedimentation, separation of solids substances and oils removal. At the end of the treatment their characteristics allow their reutilization and/or disposal with respect of the applicable normative.
Waters from the second network are treated by sedimentation, oils removal, biological treatment, pH correction and chlorination. At the end of the treatment, a portion is sent to external treatment plants for treatment and disposal, while another portion is utilized to moisten fly ashes for the abatement of their dustiness.
Main solid waste produced by the energy recovery plant include sand, bottom ashes and fly ashes which are disposed of according to the applicable normative. Bottom ashes amount to 5,000-6,500 t/y, and fly ashes to 14,000-17,500 t/y.

Compund	**Max allowed concentration** (mg/m^3)
Particulate / dust	8
Total Carbon (TOC)	8
Hydrochloric acid (HCl)	8
Hydrofluoric acid (HF)	0.8
Sulphur oxides (SO_2)	40
Nitrogen oxides (NO_x)	160

Table 6. Authorized limits for gaseous emissions

Periodical monitoring campaigns to check the acoustic emissions of the plant are also planned and carried out by the official Institutions charged of this duty.
Analogously, during plant operation measurements of the electro magnetic field are done to verify the respect of the normative limits of non ionizing radiation emissions.
Since 2006, the plant got and operates a certified ISO 14001:2004 EMAS system of environmental management.

6. Conclusion

The correct management of municipal solid waste, in a context of a sustainability concept, requires adoption of appropriate integrated systems to:
- maximize the use and utilisation of waste material and energy content;
- minimize the impact of waste on the environment.

In the Region Puglia (Apulia), SE of Italy, the "Commissariat for Environmental Emergency" was established since 1997 having, among others, the duty to develop the regional plan for municipal solid waste management in conformity with European and National regulations.

With the Commissary Decree 296/2002, as completed and adjourned by the Decree 187/2005, the Commissary approved the "Regional Solid Waste Management Plan", after introducing on 1997 and 1998 technical specifications for the mechanical-biological treatment of solid waste remaining after separation at source of selected fractions.

Basically, above mentioned Commissary Decrees, require the:

- development of "source separation" schemes with the target for 2010 of 55% of MSW separately collected to be subsequently handled for material recoveries;
- operation of Centres for the "qualification" of specific recyclable fractions deriving from above "source separation or separate collection";
- "biostabilisation" of urban waste remaining from separate collection prior to the separation of a treated wet fraction to be landfilled, or used for environmental purposes, and a dry fraction to be used for the production of refuse derived fuel.

The management plan split up the regional territory into 15 "Optimal Territorial Basins" each mainly served by treatment plant for:

- "qualification" of specific recyclable fractions deriving from "source separation or separate collection" of urban waste;
- "pre-treatment" of residual urban waste deriving from conventional "not-separate collection";
- "biostabilisation" of above pretreated waste;
- "mechanical separation" of biostabilised material into a "wet fraction" and a "dry fraction", being the former landfilled or submitted to further curing for the production of materials to be possibly reused for environmental purposes, the latter (FSC) processed for conversion into RDF;
- "landfilling" of process rejects or of untreated waste during shutdown periods for maintenance and/or emergency.

The first plant complying with requirements of the waste management regional plan was that located in Massafra, with an authorised capacity of 110,000 t/y.

Core of the plant is the biological stabilization process that takes place for 7-14 days in 13 biotunnels. The biostabilised material is then screened to obtain a "wet" (undersized) fraction and a "dry" (oversized) one. Then the dry fraction is processed to be converted into densified refuse derived fuel.

Finally, produced RDF is burnt in a dedicated power plant to recover energy. Main characteristics of the power plant are a gross electric power of about 12.5 MW_e, a net electric power of about 10.0 MW_e, a thermal power of about 49.5 MW_t, and a net efficiency of about 21%.

The plant has now cumulated almost 7 years of successful operations fully complying with limits imposed by applicable regulations.

7. Abbreviations

DRI	Dynamic Respirometric Index
EU	European Union
FSC	Treated (biostabilised) dry fraction for production of refuse derived fuel (RDF)
MSW	Municipal solid waste
OTB	Optimal territorial basin

RBD Treated (biostabilised) wet fraction for disposal in landfill
RBM Further treated (cured/matured) wet fraction for environmental utilisation
RDF Refuse derived fuel

8. Acknowledgements

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To this purpose, opinions and statements expressed in the Chapter are those of the authors and not necessarily those of above mentioned Institutions and Companies.

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5

Strength and Weakness of Municipal and Packaging Waste System in Poland

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1. Introduction

The European Union's approach to waste management is based on three principles: prevention, recycling, and reuse. The introduction to Directive 2006/12/EC of the European Parliament and the Council on Waste states that "the recovery of waste and the use of recovered materials as raw materials should be encouraged in order to conserve natural resources". According to the newest Directive 2008/98/EC on waste recovery is one of the five objectives of environment-friendly waste management. The targets for re-use and recycling of waste, which should be attained by 2020, is:

- for re-use and the recycling of waste materials such as at least paper, metal, plastic and glass from households and possibly from other origins as far as these waste streams are similar to waste from households, shall be increased to a minimum of overall 50% by weight;
- and for non hazardous construction and demolition waste: defined in category 17 05 04 in the list of waste shall be increased to a minimum of 70% by weight (EC, 2008).

Moreover, the Directive 2004/12/EC (amending Directive 94/62/EC) on packaging and packaging waste was adopted. This Directive aims to harmonize national measures in order to prevent or reduce the impact of packaging and packaging waste on the environment. Therefore the recovery (60% in 2014) and recycling (55% in 2014) targets were established and must be met by each member state. In Poland, although the recycling level for municipal waste has been increasing, it still remains at a very low level (approximately 8%). One of the reasons for this is that there are two parallel systems, which are responsible for separate collection, i.e.:

- **system for local communes,** which are responsible for management of all type of municipal waste,
- **system for entrepreneur-manufactures,** which are obliged for recovery and recycling of packaging waste.

On both systems the market conditions, i.e. relatively high cost of separate collection, which depends on the amount of collected material, the unit size of the waste material, the quality of the waste materials, changing price of waste materials, lack of education have significant influence and not allow implementing new technological solutions. As both the waste material obtained from separate collection from municipal waste and the waste packaging material should be deliver to recycling companies, the cost of their collection is a decisive factor in respect of the profitability of this process. As the cost of collection of individual waste is usually higher then bulk packaging and transport packaging waste, system for entrepreneur-manufactures is usually focus on the latter collected. The demand from Polish producers for waste materials (glass, paper, plastic) is relatively high, even the proper quality of waste with low price is required; therefore the system for entrepreneur-manufactures has higher potential to develop.

The aim of this chapter is to analyze the strength and weakness of existing systems of waste management in Poland, the assessment if the EU requirements with current systems could be achieved till 2020 and the proposal how to develop – based on best EU practice – these systems to promote both the recovery and recycling of separate collection of household waste and packaging waste.

2. The strength and weakness of local communes system

In the EU old members the planning of waste management had been developed since 1970s. In that time in most of EU new members there was central planned economy, with quite well developed system for glass reuse and metal collection. During the transformation period the waste management was not the most important subject and the waste landfilling was the most popular option. After joint the EU it was necessary to implement the EU requirements. With the EU financial support (structural funds) first it was necessary to close the ineffective landfills and then to build the system for recovery and reuse. Unfortunately this is a very slow process. Numerous economic and legal changes concerning waste management have been introduced in Poland over the last 10 years. As a result, the amount of waste deposited in landfills sites has been diminishing, dropping from over 95% a few years ago to approximately 85% last year. According to the Central Statistical Office (GUS), over 12 million Mg of waste, i.e. 319 kg per person, was generated in Poland in 2009, while about 10 million Mg (264 kg/per person) was collected, of which 8.469 million Mg was deposited in landfill sites, 0.101 million Mg was incinerated, 0.508 million Mg was subjected to biological and mechanical treatment methods, and 0.796 million Mg was segregated from mixed waste. From collected household waste 0.543 million Mg was collected separately for recycling, predominated by glass, paper and cardboard (Fig. 1 and 2).

Segregated collection has been increasing, though very slowly, mainly for economic reasons such as the fact that the price of the material separated from the waste remains low, and therefore there is not interest of implementing new technological solutions. As one of the aims set out in, for example, the National Waste Management Plan 2010 and the ecological policy, is to increase the recovery or recycling of waste material from household waste (glass, paper, metal) from the current level of 8%, to 50% of the overall quantity by 2020, new solutions should therefore be developed for the promotion of both separate collection and the segregation of material from mixed waste.

In some communes, the selective collection of waste is financed from budget sources, as the communes are responsible for keeping their region clean. A company, chosen by means of open bidding, empties the special selective waste collection containers, known as 'bells'. In 2004, the average total cost, including investment, for segregated collection in communes

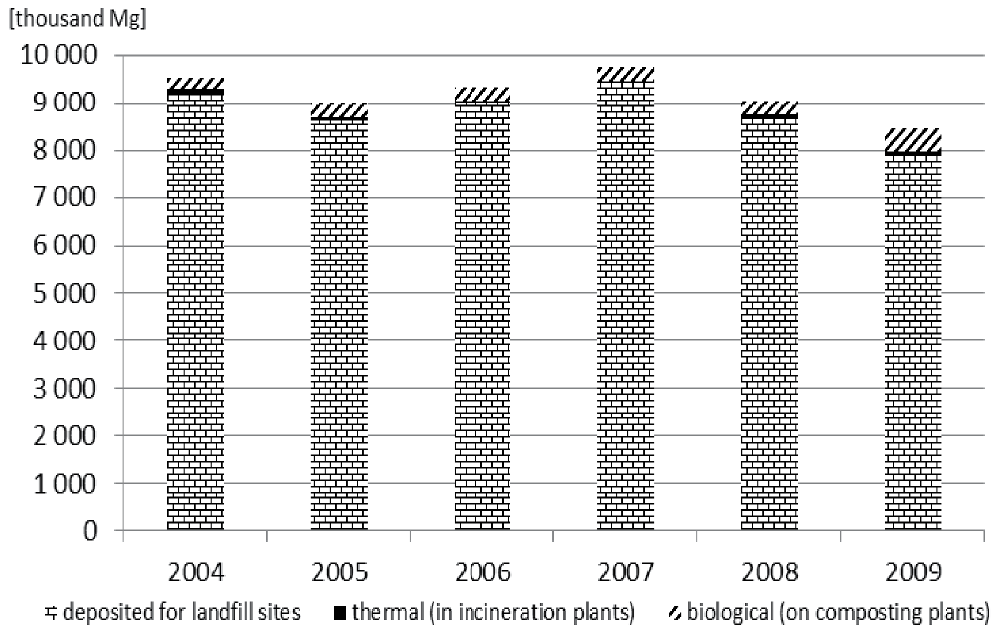

Fig. 1. Municipal solid waste managed in 2004-2009

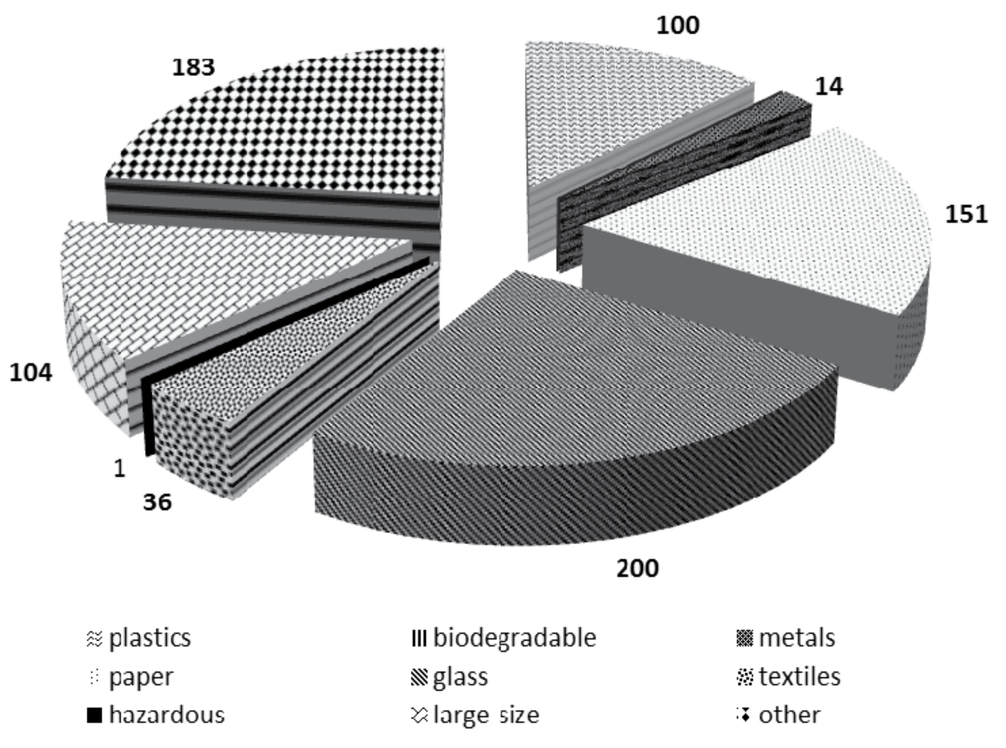

Fig. 2. Segregated municipal waste collection in 2009 [thousand Mg per year]

varied from PLN 235/Mg (glass) to over PLN 1.160/Mg (aluminium) (Poskrobko, 2005). Similar cost levels for waste collection were obtained in various towns in 2006, e.g. in Tarnów, the average cost was over PLN 600/Mg (Report, 2007). Most communes concluded that separate collection is not in the least profitable, with every PLN 1 received for the material obtained having incurred a collection cost of PLN 4. In order to reduce collection costs, the collection companies have now introduced a bag system. The cost of collecting paper in 120 litre bags was PLN 60/Mg, with plastic costing PLN 200/Mg and glass, in 80 litre bags, costing PLN 27/Mg (OGIR, 2008). An even more effective system proved to be the provision of one bag for mixed paper, plastic and glass waste. This solution made it possible to increase the amount of waste for recycling, and to cover the costs of collection for some types of waste material; for example, the price of waste paper might then be approximately PLN 100/Mg. However, even if some income could be earned from the sale of paper, plastic materials, glass and aluminium tins for recycling, it would not reach a level permitting investment in, and the development of, such an operation.

However, this system is fully dependent on market conditions, which are changeable. Therefore, other incentives for promoting recovery should thus be implemented, for instance, a system of awards for individual 'collectors', educational measures, or the seeking of financial support from Structural Funds for new technological solutions, and so forth.

The strength and weakness of local communes system is presented in table 1.

Even there are some improvements in waste management in Polish regions, it is important to elaborate in regional plans a conceptual model, which can promote waste recycling and recovery including regional conditions. Such model was proposed e.g. in South East England. The model was developed for the recycling chain for each priority materials. The five stages model has been analyzed and it included: collection, pre-processing (sorting/segregation), densification (volume/size reduction), reprocessing (conversion ratio into raw material) and fabrication (produce/product). This structure has been proposed to each priority material to establish the size and distribution of capacity at each point in the chain. It is recognized that some routes combine steps in the chain. For example newspaper recycling to newsprint may go direct from collection to reprocessing and fabrication (Potter, 2006). Based on such model the regional plans should set realistic targets for all form of waste. It is particularly important that communes should work together in the area where there are opportunities to achieve better value for money and to achieve sustainable waste management.

Moreover, for the evaluation of environmental impact of waste processes or systems one of the most respected, popular and widely used in the EU method is LCA (Life Cycle Assessment). The method has been seized, inter alia, to develop The Strategic Environmental Impact Assessment for the National Waste Management Plan in the Netherlands and Strategic Environmental Impact Assessment for the Waste Management Plan of the region of Liguria in Italy. Worldwide, there are many programs that use the LCA for supporting modelling of waste systems as well as evaluating their impact on the environment, i.e. IWM-2 (*Integrated Waste Management II*), WRATE (*The Waste Resources Assessment Tool for Environment*), TRACI (*Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts*), EASEWASTE (*Environmental Assessment of Solid Waste Systems and Technologies*), ORWARE (*Organic Waste Research*), WISARD (*Waste – Integrated Systems for Assessment of Recovery and Disposal*), and more general software as SimaPro and GaBi. These programs are used to evaluate both the existing as well as the modelling of new waste management systems and to determine the environmental benefits of their modernization.

Introduction of such assessment could be beneficial also for new members, especially, as some proposals have already been done by JRC, Ispra (Koneczny et al., 2007).

strength	weakness
the planning system – based on the EU experience – has been introduced including aims, tasks and costs of its realization	lack of proper legal regulation which allow communes to manage the waste as the owner of waste. The owner of waste could be transport companies or owners of waste management facilities, i.e. sorting installation or landfills
communes started to cooperate with each other creating larger organization system for separate collection	communes are relatively small, therefore the management of waste is dispersed, and as a result there are not enough specialists responsible for waste management, planning and reporting in communes
the existence of environmental fee and fines system, which are separate from tax system	lack of common scheme for collecting and recording data for type of waste, methods of recovery and recycling, etc.
small progress in separate collection has been achieved in last years	there is not regional system of waste management, which should be connected with the regional conditions i.e. if there is a glass factory in the region the system of glass collection should be promoted
the separate collection system (i.e. bells or bags) is available for about 50% inhabitants in some regions in Poland, but not all inhabitants are used it	relatively low cost of landfilling (including environmental fee) compared to other methods
availability of financial support for new installations and education from EU –fund as well National Fund of Environmental Protection and Water Management	lack of systematic education as well lack of education provided by individual regions
	lack of economic encouragement for privet investors for development of separate collection, therefore there are only few sorting plants where waste from individual household should be cleaned
	there are not legal instruments to force to achieve the indicated in local and regional plans level for separate collection

Table 1. The strength and weakness of local communes system in Poland

3. The strength, weakness of entrepreneur – manufactures system (packaging waste)

Poland has already adopted the majority of the EU regulations, e.g. the Directive 2004/12/EC (amending Directive 94/62/EC) on packaging and packaging waste, which imposes the obligation of adopting specified packaging waste recovery and recycling levels on Member States. The Directive was introduced into Polish law in 2001, and updated over the course of the following years. The entrepreneur-manufactures or importers of packed materials were obliged to attain the appropriate percentage level for mass of the packaging waste towards the implemented packaging mass. The legislation permits the delegation of this obligation to a recovery organization. If they fail to attain the statutory level of recycling, they are obliged to pay a product fee for the difference between the required and the achieved level of recovery and/or recycling, expressed in product weight or quantity[1]. The fees are imposed on entrepreneur-manufactures or importers of packaging materials. The system is very complicated, as the duty imposed on an individual company for different types of packaging material, and not on the total tonnage of packaging material, can be met by company itself, or by a recovery organization. The product fee is in correlation with the collection costs, but the cost of collection from an industrial source (bulky packaging waste) is several time lower than from individual one. In 2008, the product fee varied from PLN 0.26/kg for glass to PLN 2.37/kg for plastic. In general, being higher than the price, which can be obtained for material separated from municipal waste (Kulczycka & Kowalski, 2010). The system seems to be very effective, given that official statistics suggest that the required level of recycling for all types of packaging material was not only achieved, but, in a number of years, was even significantly exceeded (tab. 2). The very high level of recycling in 2004-2006 presented here was mainly due to the system of classification introduced by the Ministry of the Environment, whereby if required annual recovery and recycling levels excess 100%, were carried forward to the report for the next year. This was amended in 2007 and from then on reported recovery and recycling levels have not included the aforementioned surplus (GUS, 2009).

Year	2003		2004		2005		2006		2007		2008		2009		2014
	A	R	A	R	A	R	A	R	A	R	A	R	A	R	R
Plastics	16.8	10.0	22.4	14.0	30.3	18.0	36.9	22.0	28.0	25.0	23.9	16.0	21.5	17.0	22.5
Aluminum	27.1	20.0	33.3	25.0	86.7	30.0	110.4	35.0	82.0	40.0	60.9	41.0	64.2	43.0	50.0
Steel	14.4	8.0	17.3	11.0	23.4	14.0	34.1	18.0	21.2	20.0	26.5	25.0	33.6	29.0	50.0
Paper	52.9	38.0	57.0	39.0	65.4	42.0	85.6	45.0	69.1	48.0	67.2	49.0	50.9	50.0	60.0
Glass	20.4	16.0	31.2	22.0	38.4	29.0	48.0	35.0	39.7	40.0	43.9	39.0	41.9	430	60.0
Natural materials	9.0	7.0	19.4	9.0	47.2	11.0	73.4	13.0	47.8	15.0	26.3	15.0	23.1	15.0	15.0
Multi material	13.5	–	14.2	–	22.5	–	–	–	–	–	–	-	-	-	-

A – achieved; R – required

Table 2. Required and attained recycling and recovery levels for packaging material in 2003-2009 and required level for 2014 (in percent %)
Source: GUS

[1] The Minister of the Environment's Regulation of 14 June 2007 on annual levels of recovery and recycling of packaging and post-usage waste (O. J. No. 109 item 752) stipulated the required level of recovery and recycling.

In Poland the quantity of packaging product launched on to the market has increased from approximately 3.1 million Mg in 2005 to about 3.8 million Mg in 2009, and officially about 37% of packaging waste undergoes recycling process. At over 43% the main packaging material is paper, namely packaging made from corrugated and solid cardboard and glass (fig. 3). Bulk packaging and transport packaging waste are predominant here, as they are easy to localize because they occur in the trade and industry sectors. Glass packaging holds second place owing to the extensive production of the disposable packaging that facilitates the disposal of packaging waste.

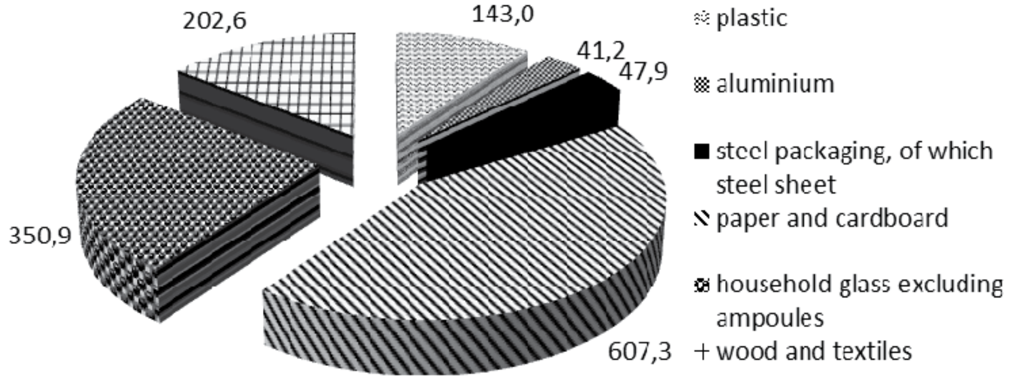

Fig. 3. Recycled packaging waste Poland in 2009 [thousand Mg]
Source: GUS

In spite of possessing the higher capacity for recycling especially for plastics and glass the owners of the recycling companies are unable to bear the high costs of selective collection. Meanwhile, the entrepreneur-manufacturers limit themselves to the statutory recovery and recycling levels to which they are bound. Product fee sanctions can be imposed on the entrepreneur-manufacturers only in cases where these levels are not met; at the same time, most of them are able to achieve this level owing to the fact that they can fulfil their obligations by means of Recovery Organization on the free market to buy so-called 'receipts' (there are about 40 of such Recovery Organizations on Polish market). An organization introducing packaging and products on to the market can buy the appropriate amount of 'virtual receipts'; corresponding to the quantity it should meet in order to fulfil its recovery and recycling obligations. The financial resources for fulfilling this obligation are known as a 'recycling payment'. When the act initially came in to force, these recycling payments were high, though they did not exceed 50% of the product payment. However, as the system was not watertight, some 'virtual receipts' were incorporated in the relevant calculations several times, and the price of the recovery payment thus dropped significantly. As a result about producers and importers of packaging waste paid 5 million PLN/year as a product fee, whereas about 60 million PLN/year to Recovery Organizations in last years, whereas the real cost of collection of 1,5 million Mg of packaging waste was estimated on 300 million PLN (Kawczyński, 2009).

The existing entrepreneur-manufactures system is presented on Fig. 4.

The revenues from product fees are distributed (according to the Act on requirements for entrepreneurs with respect to management of some wastes and product and deposit fees-consolidated text O. J. 2007, no. 90 item 607) to:

- Marshal Offices receive 100% of revenues,
- Marshall Offices keep 2%, while 98% is transferred to the National Environmental Protection and Water Management Fund (NFOSiGW),
- NFOSiGW keeps 30% of revenues, while 70% is transferred to Voivodship Environmental Protection and Water Management Funds (WFOSiGW), which transfer all the resources to the Communes Office (as income of the communes, fig. 5),
- Redistribution of funds from product payments for packages, based on the indicator of the quantity of package waste assigned for recovery and recycling, causes the funds from the voivodships which gain high revenues from product payments to be transferred to the voivodships which gain low revenues (GUS).

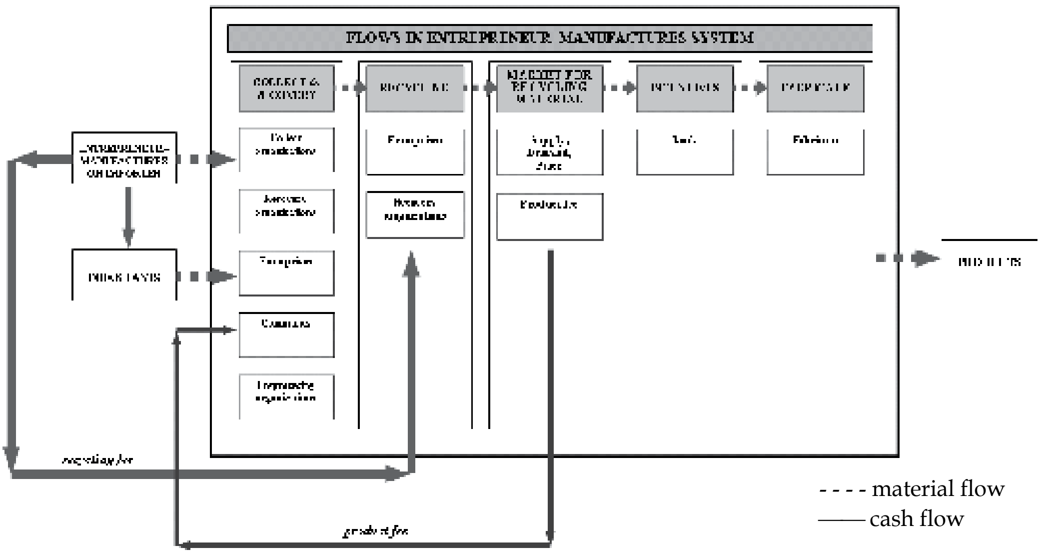

Fig. 4. The flow of packing waste in entrepreneur-manufactures system

In year 2004-2009 the total value of product fee paid by Marshals' Office varied from 1,5-4,2 million PLN, with significant drop in last 2 years (tab. 3). In 2009 the product fee paid to Marshal Offices amounted to 1,5 million PLN, whereas due product fee 4,5 million PLN (tab. 3). The value of due product fee is at least two times higher than regular product fee, as many entrepreneur-manufacturers paid with delay (then the interest for delay is added). The sum of product fee and additional fee transferred from Marshals' Office to NFOŚiGW differs from the value of receipts of NFOŚiGW as the presented in table 3 data were calculated during the calendar year, whereas the transfer of fee Marshals' Office to NFOŚiGW was till the end of April (fig. 5).

Communes also collect packaging waste, but they not always proper divided municipal waste from packaging waste. It creates a lot of problems in waste classification, quite often double counting (as entrepreneur-manufacturers or recovery organization can re-collect the packaging waste from communes) and reporting. The strength and weakness of entrepreneur-manufactures system is presented in table 3.

A proposal for new regulations for the management of packaging waste has been put forward in order to attempt to solve these problems. However, it does not propose any structural changes, such as taking into account the ratio of tones of *packaging to tones of*

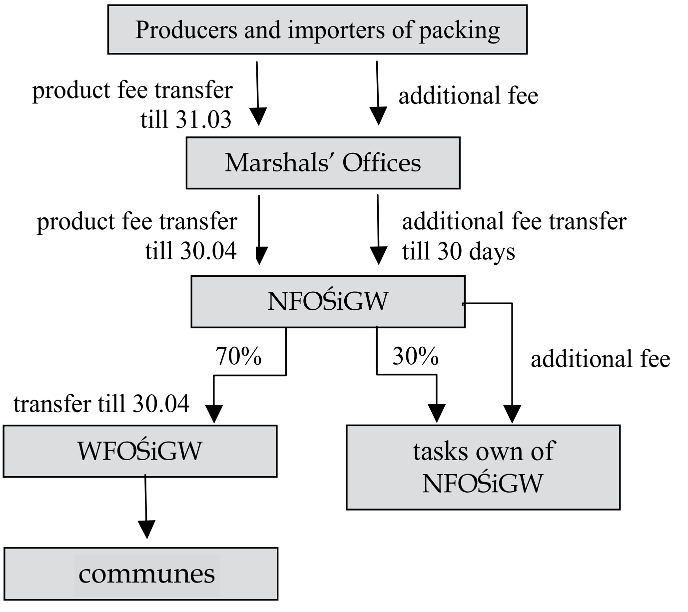

Fig. 5. Redistribution of product fee in Poland. Source: GUS

product (sold by weight), or the share of non-recyclable packaging within the total tonnage (e.g. total non-recyclable packaging as a percent of total packaging sold) or introducing correlation between packaging size and product fee. Some incentives should be implemented for companies, who introduced the packaging from recycling materials, for example for these packaging materials the product fee should be suspended.

Taking into account the best practice from other EU regions (Boag, 2010) the following idea may be taken into consideration:

- the objective of the system should be to discourage producers from generating waste in the first place,
- if the packaging waste are placed on the market the evidence system should be verified gathering information from different sources, i.e. associations or NGO,
- the recycling companies should be accredited or by the National Fund of Environmental Protection and Water Management or the Ministry of the Environment, i.e. in Scotland there are 31 SEPA accredited re-processors and exporters of packaging waste, and 415 re-processors and exporters for the UK in total,
- the entrepreneur-manufactures or importers of packed materials should be registered in the system special created for them, for example in Scotland producers for registration can join a compliance scheme (private business ensures recycling evidence on behalf of producers), or to register with the relevant environment agency and obtain the evidence of recycling. There are 5 compliance schemes, which have registered with SEPA (120 members in total). In the UK as a whole there are 48 Schemes with a total of 6487 members. Whereas 76 companies have directly registered with SEPA in Scotland and 535 for the UK,
- the regional or national web-based database with on-line submission, both from producers and re-processors should be created, i.e. in the UK the National Packaging Waste Data base was established in 2005 and it is supported by the UK environment

agencies, government and companies obligated by the packaging regulations including re-processors, exporters and compliance schemes. In Poland the regional data based was also created, but it includes information concerning mainly municipal solid waste, i.e.

- the amount and type of produced waste and the ways of its management,
- the registry of the issued decisions regarding waste production and management,
- the waste management plans,
- installations that are used in order to reclaim and neutralize waste with separation of the landfills and installations for thermal transformation of waste (Góralczyk et al., 2008).

Product fee / Year	2004	2005	2006	2007	2008	2009
Value of total product fee as well as additional product fee paid to Marshals' Offices	3 261,5	2 799,0	4 217,5	3 357,2	1 925,1	1 500,5
Value of due product fee as well as additional product fee paid to Marshals' Offices	n.a.	4 306,7	4 604,6	9 103,6	6 571,6	4 471,6
Receipts from Marshals' Office for the NFOSiGW	7 097,4	9 545,0	6 116,9	13 819,8	11 441,3	7 162,8

Table 3. Value of product fee and its distribution (thousand PLN)
Source: GUS

strength	weakness
clear defined responsibility on entrepreneur-manufactures or importers for collection and recycling the required level of packaging waste	system is complicated, as the duty are imposed for different types of packaging materials, and not on the total tonnage of packaging material
there are legal and financial instruments to fulfil the obligation	lack of clear rules for documents and information flow in the system what allows to create of 'virtual receipts'
significant increase in recovery and recycling of packaging waste in Poland	not coherent classification of waste (as communes not always divided municipal waste from packaging waste)
income from product fee are dedicated mainly for improvement of the waste system including educations and promotions	double counting - as entrepreneur-manufacturers or recovery organization can re-collect the packaging waste from communes
	lack of control on recycling organization and proper reporting
	lack of economic instrument or other system for encouragement for recovery or recycling
	the system does not encourage to the prevention of waste

Table 4. The strength and weakness of entrepreneur-manufactures system in Poland

4. Conclusions

1. The simultaneous introduction of an effective instrument promoting the recovery of waste from both municipal waste and packaging waste in new EU members is difficult, as these two systems deliver their final products to the same recycling companies and are thus forced to compete on the market.
2. Despite the high capacity of recycling companies in Poland, only about 50% of the packaging waste introduced into Polish market undergoes the utilization process. Although they possess the recycling capacity for plastics and glass, the owners of the recycling companies are unable to bear the high costs of the selective collection.
3. The collection costs for the individual packaging waste obtained from municipal waste is much higher than the costs of collection of transport, or cumulative packaging.
4. Owing to its lack of economic efficiency (high collection costs), the communes system is support from local authorities in the majority of cases. However, such a system cannot be applied in the long term, as the local authorities have no specific funds allocated to such operations.
5. The entrepreneur-manufactures system introduced in Poland is complicated as it was addressed to individual entrepreneurs manufacturing for each type of packaging waste, rather than addressing the total tonnage of packaging waste introduced on to the market.
6. There are no other measures (the ratio of tones of *packaging to tones of product* sold, or the share of non-recyclable packaging in total tonnage) then the tonnage of packaging waste introduced on to the market for the assessment of entrepreneur-manufactures system. As a result, the obligation is fulfilled by either packaging or cumulative (bulk and transport) packaging, and not by individual packaging.
7. Recovery and recycling obligations can be met by various entities, i.e. the company itself or a recovery organization; a failure to meet the obligation on the part these entities results in the requirement to pay a product fee. This solution gave rise to the market for so-called 'receipts'. Corresponding to the quantities designated for recovery and recycling. Entrepreneurs from recovery organizations buy them, the latter being responsible for mediation between companies and the processors of the obligations. There are therefore no effective incentives for entrepreneurs to decrease the packaging waste tonnage as, owing to competition on the market, the cost of the 'receipts' dropped. As it stands, the act also fails to encourage entrepreneurs to co-finance collection, recovery and recycling. This obligation was placed on residents, communes and recycling companies.
8. The education system and especially, the introduction of marketing instruments for the promotion of segregated collection at 'source' is insufficient.
9. The instruments for prevention of generating waste or eco-designing is not well developed.
10. It is necessary to introduce the other than based on market solutions incentives, i.e. support for developing new technological solutions.

5. Acknowledgements

This research was supported by the project Assessment of the municipal waste management system and methods for its evaluation N R14 0016 04 from National Centre for Research and Development.

6. Abbreviations

EASEWASTE – Environmental Assessment of Solid Waste Systems and Technologies,
EU – European Union,
GUS – Central Statistical Office,
IWM-2 – Integrated Waste Management II,
LCA – Life Cycle Assessment,
ORWARE – Organic Waste Research,
PLN – Polish zloty, is the name of Polish currency,
TRACI – Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts,
WFOSiGW - Voivodship Environmental Protection and Water Management Funds
WISARD – Waste Integrated Systems for Assessment of Recovery and Disposal,
WRATE – The Waste Resources Assessment Tool for Environment.

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Management of Municipal Solid Wastes: A Case Study in Limpopo Province, South Africa

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1. Introduction

Wastes are inevitable part of human activity. The problems associated with waste can be traced back to the very beginning of civilization, when humans gathered in communities (Priestly, 1968). Wastes generated then were contained and disposed of by natural processes. However, as population increased and villages grew into towns and then into cities, the amount of waste generated increased. Consequently, wastes were dumped indiscriminately into waterways, empty lands and access roads. The appalling conditions gave rise to epidemics like the ''Black Plague'' that destroyed large population of Europe in the 14th century (Priestly, 1968). Similar conditions were also experienced in the other continents.
The industrial revolution that took place in Europe in the 19th century marked a turning point in waste management. It brought with it, among other things, migration of people from rural areas to towns and cities in search of jobs. The resulting concentration of people in towns and cities gave rise to alarming proportion of wastes being dumped in the streets and waterways. Legislations were passed by the governments of the day in order to curb the indiscriminate dumping of waste. Progress was slow until a positive link was established between vermin invested wastes and the spread of disease. The discovery of pathogens as the agents of diseases that for centuries had been the scourge of mankind, paved the way to the modern sanitary practice.
The modern practices of waste management such as open dumping, incineration (burning), composting and landfill (burial), can be traced back to early civilization. However, the practices were conducted haphazardly and specific to particular cultures and traditions. The quest for a cleaner environment has introduced the modern systematic management approach of storage, collection and disposal of waste. Countries all over the world have acts that provide for the removal by the local authorities, on specified days, accumulated wastes from premises. The waste is normally placed into removal receptacles, bags and bins, for easier removal. This rapid revolution of waste management that started in the developed countries has spread to the developing countries, particularly the more affluent areas. The

advances in technology have also contributed immensely to the common practices today of waste management.

Fifteen years back, waste management was not regarded as a national priority issue in South Africa. The waste management practice that took place before 1994 focused mainly on waste disposal. The low emphasis that was accorded to waste management has resulted in waste impacting negatively on South African environment and on human health. In 1999 South Africa adopted the National Waste Management Strategy. This strategy outlines the goals to address waste management in the country. The entire strategy is based on a 3R (Re-use, Re-duce, Re-cycle) to improve the quality of the environmental resources affected by uncontrolled and uncoordinated waste management.

South Africa realized the impact of waste on the environment and a policy referred to as ''Integrated Pollution and Waste Management Policy (IPWM) of 2000 was established (DEAT, 2000). The policy outlined goals to be achieved through the National Waste Management Strategy (NWMS) of 2001 (DEAT, 2001) and focuses on different key elements of integrated waste management planning, waste information system, general waste collection, waste minimization, recycling, waste treatment and disposal, capacity building, education and awareness. The key objective of the NWMS is to reduce waste generation and environmental impact of all forms of waste and to ensure that the health of the people and the quality of the environment are no longer affected by uncontrolled and uncoordinated waste management. This chapter looks at the historical perspective, best practices and waste management in Polokwane City.

The Limpopo Province has six district municipalities of which Polokwane city is found within the Capricon District municipality. Polokwane city previously known as Pietersburg is about 270 km to the northeast of Tswane (Pretoria) City. Polokwane City lies between 23^{0} 54′ 59″ S latitude and 29^{0} 27′ 00″ E longitudes (Fig. 1).

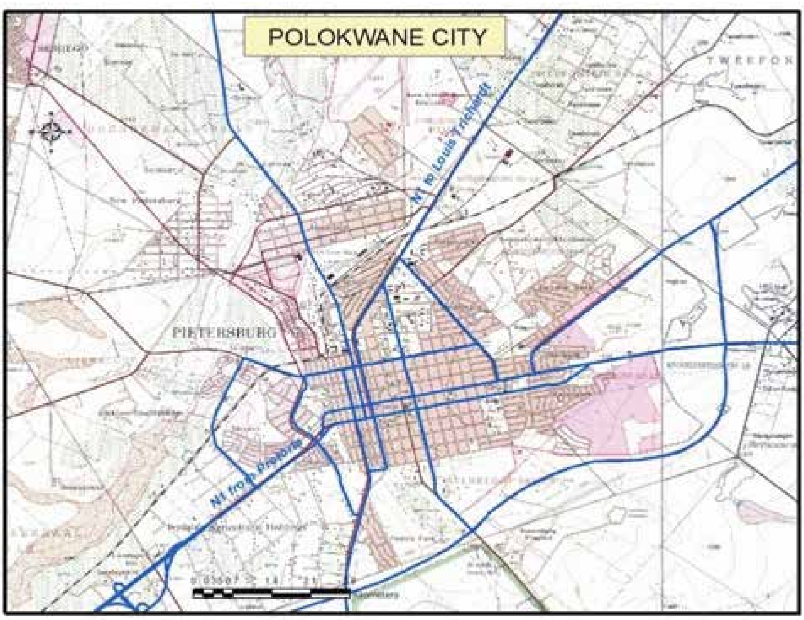

Fig. 1. Location map of Polokwane city (Source: Spot 5, Satellite image CSIR, 2007).

Polokwane area generally has rain in Summer and dry in Winter. Land use in Polokwane city falls under the following categories: business and retail, industrial, community services, recreation and tourism, residential, agricultural and nature conservation areas.

2. Sources of waste

There are six general sources of waste generation, namely; domestic, commercial, industrial, agricultural, institutional and natural:

- Households are the highest producers of domestic waste. Domestic waste includes, among others, paper and cartons, plastics, glass, left over food, cans.
- The main agents of commercial waste producers are stores, business premises, markets and restaurants.
- Industrial waste refers to wastes such as construction and demolition debris and food processing outlets.
- Agricultural wastes refer to the waste outcomes from dairy and poultry farms, livestock and other agricultural activities like vegetation cultivation. Most of the agricultural wastes contain biodegradable components.
- In case of institutional wastes, major producers are schools, offices and banks. This type of waste contains paper and cartons.
- Natural waste consists of leaves, tree branches, seeds and carcasses of animals.

3. Solid waste hierarchy

The waste management hierarchy has been adopted by most industrialized countries as the menu for developing waste management strategies. According to Seadon (2006), many programmes have adopted waste management hierarchy to address solid waste, for example, New Zealand's Local Government Act Amendment No. 4 (1996) defines hierarchy as ''reduction, reuse, recycling, recovery, treatment and disposal'' with desirability of decreasing down the hierarchy.

In Singapore, the hierarchy is based on waste minimization (reduce, reuse, and recycle-3R) followed by incineration and landfill. Land is very scarce in this country and this has resulted in incineration as the most preferred method of treatment (Bai and Suntanto, 2001). The United States Environmental Protection Agency (USEPA, 2006) has ranked the most environmentally sound strategies for Municipal Solid Waste (MSW) as source reduction (including reuse) the most preferred method, followed by recycling and composting, and, lastly, disposal in combustion facilities and landfills.

South Africa has adopted solid hierarchy that puts waste prevention as a priority followed by waste minimization through cleaner production (Table 1). The second preferred method is recycling which entails re-use, recovery and composting of waste generated. The third method is treatment process, for example, incineration of waste prior to disposal. Landfill disposal is regarded as the last resort for all waste that remains from the other three methods.

Integrated waste management (IWM) in its simplest forms incorporates the waste management hierarchy by considering direct impacts (transportation, collection, treatment and disposal of waste) and indirect impacts (use of waste material and energy outside the waste management system). IWM is also a process of change that gradually brings in the management of wastes from all media (solid, liquid, and gas) (Seadon, 2006).

WASTEHIERARCHY		
Focus	Activity	Order of aplicación
CleanerProduction	Prevention	
	Minimisation	
Recycling	Re-Use	
	Recovery	
	Composting	
Treatment	Physical	
	Chemical	
	Destruction	
Disposal	Landfill	

Table 1. Solid waste hierarchy adopted by South Africa (DEAT, 2001)

4. Components of solid waste management system

4.1 Waste generation

According to the 1999 State of Environmental Report for South Africa (DEAT, 1999), the country generates over 42 million m^3 of solid waste every year. This is about 0.7 kg per person per day, compared to 0.73 kg, 0.87 kg and 0.3 kg for the UK, Singapore and Nepal respectively. In 2001, the amount of general waste produced throughout South Africa was reported to increase annually due to population growth, economic growth and unsustainable lifestyles. In South Africa, disparities in the volumes of waste generation between higher income groups and lower income groups exist. In general, the higher income groups generate more waste per capital (2.7 m^3/per capital/annum) than the lower groups (0.2 m^3/capital per annum) (DWAF, 1998).

The State of Environmental Report (2003) compiled for the City of Johannesburg indicated that the City with its all regions and population of 2,982, 033, generates a total of 1,560,400 tonnes of waste per annum. There has also been prediction of 1,700,000 tonnes of waste by 2010. The waste generated in this City is as follows: 23% commercial activities, 10 % industrial and 67 % household wastes. Households generate a total of 889,665 tons a year and this is also expected to increase to 1,079 055 tons a year, based on the economic development growth of the City.

High income earners currently generate on average 1.3-1.6 kg of waste per day. Middle income earners generate 0.7-1 kg per day and lower income group generates 0.35-0.6 kg of waste per day (City of Johannesburg Report, 2003). The City of Johannesburg recycles 6-8% of waste, and with the prediction in waste generation increase, it will therefore be necessary for the City of Johannesburg to increase the levels of recycling.

Waste Collection and Transportation

Solid waste collection and transportation are often the most costly components of a local waste management system. A study conducted in Ghana outlined the collection system and design aimed at significant cost savings focusing on the utilization of waste containers fitted with level sensors and wireless communication equipment, which alerts the waste collector's access to information on the status of each container (Amponsah and Salhi, 2004). Transporting of waste can be done either using the collector trucks, canal or rail. Transportation through the canal has been regarded as an option that can save fuel, although it has not yet been practiced anywhere since there are no facilities for canal (Kulcar, 1996).

The collection system is uniform in most countries, wherein individual households place their daily refuse into a container nearby, then the refuse is collected and delivered to the waste collection point or disposal site (Fig. 2). The challenging aspect has been source-separation of household waste which is one of the key elements of the integrated waste management system, and which has not been successfully implemented in municipalities of many countries such as China (Hui et al., 2005).

Fig. 2. Waste collection at residential areas of Polokwane City.

Population size and density, the area size, the quantity of household generated solid waste, the collection distance, the geometric design of streets and roadways, and the level of area traffic congestion all have a pronounced impact on the cost of collection and transport of solid waste to the landfills. Difficulties in populated areas are pushing disposal facilities away from waste sources and increasing the costs of transporting wastes. Transfer stations can potentially reduce the costs of transporting wastes (Koushki et al., 2004).

According to the survey conducted in Sri Lanka, it has been revealed that 50% of local authorities collect less than 2 tons of waste per day and only 24% of households in the Southern Province of Sri Lanka have access to waste collection, hence rural areas have a waste collection of less than 2% (Vidanaarachchi et al., 2005).

4.2 Waste disposal

The nature and procedures for waste disposal vary from country to country. In South Africa, there are about 540 landfill sites of which 61% have permits, however, there could be 15,000 landfill sites including communal sites in the country (DEAT, 1999*)*. The 5 million tons of waste produced every year, only 5% is disposed of in designated sites, thus most waste in South Africa is disposed in environmentally unsafe sites.

The State of Environmental Report for the City of Cape Town (2003) outlines the expanding economy, increasing population and visitors as contributing factors to the increased waste generation rates in the city. This has contributed to 7% increase in waste landfilled between

2001 and 2002, which is far in excess of 2% population growth. This report reflects that 90% of waste generated in the City of Cape Town is landfilled. In 2002 a total of 1,722,807 tonnes of waste was disposed at the six landfills and this showed an increase of 7.3% as compared to 1,596,000 tonnes disposed in 2001, which was an increase of 6.5% from 1,493,000 tones generated in 2000.

Waste landfilled consists of 30% household waste, 15% sewage sludge and 55% industrial and commercial waste. The amount of waste recycled in 2002 was 2% as a result of informal salvaging activities. This percentage of recycling will increase as the city is currently busy with the material recovery facility plans. The City of Johannesburg's State of Environment Report (2003) shows that there are six landfills with a total of 1,560,400 tonnes of waste disposed annually, which shows that there is more waste disposed to waste disposal sites in the City of Cape Town as compared to the City of Johannesburg.

5. Strategies to improve solid waste management system

5.1 Waste minimisation and recycling

It has been observed that many countries such as the USA have been engaged in waste minimization strategies through waste recycling. This has been confirmed by the statistical records from 1960 to 2005, wherein recycling increased from 6,4% to 32,6%. According to the information on Table 2, recycling has diverted almost 82 million tons of recyclable material away from disposal. Typical materials recycled include batteries recycled at a rate of 99%, paper and paperboard at 52% and yard trimmings at 62%. These materials have been recycled through the curbside programs, drop off centers, buy-back programs and deposit systems.

No	Year	Total waste in tons	Total waste recyclable(tons)	Recycling (%)rate
1	1960	88,1 million	5,6 million	6,4
2	1970	121.1 million	8 million	6,6
3	1980	151,6 million	14,5 million	9,6
4	1990	205 million	33,2 million	16,2
5	2005	251 million	82 million	32,6

Source : www.epa.gov.msw/facts (2006)

Table 2. Municipal solid waste recycling rates in the USA (1960-2005)

Other countries like England and Wales, have a strategy for waste management referred to as Waste Strategy 2000, which also introduced statutory targets of waste reduction through recycling as follows: 20% of household waste 2003/4; 30% of household waste 2005/6; 30% of household waste by 2010 and 33% of household waste by 2015. These reduction targets were also applicable to biodegradable waste to 35% reduction. Oxfordshire's residents produce 300,000 tonnes of household waste per year. In 2001/2, 17% of this waste was recycled or composted and 83% was landfilled. The targets set out puts the municipalities under pressure of having to double the quantities of waste currently recycled.

The Taiwan government introduced a restriction programme for plastic bags and disposable dishes use as a way of altering the throw away consumer habits of the public. This programme was aimed at encouraging the businesses to introduce re-usable shopping bags and dishes. The target was to reduce the amount of plastic bags by 20,000 tonnes annually, which had an effect since 31% reduction rate, was achieved. The same applies to disposable dishes where consumption was 12,000 tons and reduction rate of 28% was achieved.

South Africa developed a national waste management strategy in 1997 which outlines the different action plans that include waste minimisation and recycling. This action plan resulted in the formulation of guideline on recycling of solid waste for the municipalities to use when implementing recycling programmes in their areas

Recycling in South Africa has so far focused mainly on paper, glass, plastics and metals. Well established companies have been involved in recycling in order to reduce the utilization of natural raw materials as resources in the production systems. Recycling plays an important role in the reduction of landfill space. For example, 1 tonne of paper waste occupies 3 m^3 of landfill space. The following facts represent a brief state of recycling in South Africa (PACSA, 2002):

- In 1999 it was reported that the paper industry recycled 720,000 tonnes per annum which represents 38% of paper produced and an increase from 29% in 1984. Out of the 3% recycled waste in that year, only 2% was from domestic waste. Almost each and every type of paper in South Africa has a recycling content. For example, newspaper contains 25% recycled paper, cardboxes 50%.
- Total plastics collected in South Africa were 113,000 tonnes which was 13%. This quantity had resulted in placing South Africa in the fore front in plastic recycling industry world-wide.
- Glass collection has grown from 54,370 tonnes in 1986 to 104,550 tonnes in 1999. The total tonnage produced in 1999 was 520,000 tonnes, thus 20% was recycled.
- There were 32,130 tonnes of returnable bottles that were collected in 1999 from South African Brewery (SAB) and Coca Cola Company as bottles that reached the end of life. The quantities of bottles increased as a result of change-over from 1 litre bottles to 1.25 litre bottles, which resulted in 8,000 tonnes of bottles collected.
- Steel beverage cans have a high recovery rate in South Africa as it has grown from 18% in 1992 to 63% in 1998. These increases have also been affected by the subsidies offered by "Collect a can" for collection system. Based on assessment made on the rate of collection for different recyclables, materials without subsidies like glass always had the lowest recovery rate.

Currently, in South Africa, the statistics presented by Packaging Council of South Africa (PACSA, 2002) shows that recycling is increasing enormously from time to time with an increase of above 168 % over a period of 18 years (Table 3).

Material	1984 (tons)	2000 (tons)	2002 (tons)	Increase (tons)
Paper and Board	365 000	770 000	922 000	557 000
Plastics	37 000	133 000	150 000	113 000
Metal	34 000	121 000	119 000	85 000
Glass	50 000	102 500	114 000	64 000
Total	486 000	1 126 000	1 305 000	819 000

Table 3. Total recyclable materials collected in South Africa (1984-2002)

5.2 Source reduction

Source reduction involves altering the design, manufacture, or use of products and materials to reduce the amount and toxicity of what gets thrown away. Source reduction can be a successful method of reducing waste generation. Practices such as glass recycling, backyard composting, two-sided copying of paper, and transport packaging reduction by industry have yielded substantial benefits through source reduction. Source reduction has many environmental benefits. It prevents emissions of many greenhouse gases, reduces pollutants, saves energy, conserves resources, and reduces the need for new landfills and incinerators.

More than 55 million tons of municipal solid waste were source reduced in the United States in 2000, and this comprised 28% containers and packaging materials, 17% non-durable goods (newspapers, clothing) , 10% durable goods (appliances, furniture, tires), 45% other MSW (yard trimmings, food scraps) (www.epa.gov.msw/facts, 2006). Most countries have developed strategies aimed at reducing waste generation by addressing waste from the source.

Polokwane Declaration on Zero Waste by 2022 was agreed upon at a meeting held in Polokwane city in 2000 so as to address the problems of waste in the country. This declaration was based on the urgent need to reduce, re-use and recycle waste in order to protect the environment and the waste management system which promotes effective waste reduction. The goal of this declaration was to reduce waste generation and disposal by 50% and 25% respectively by 2012 and develop a zero waste plan by 2022. The South African Government developed a National Waste Management Strategy to address waste management aspects including the zero waste plan as envisaged.

Other initiatives taken by the South African Government is the plastic bag agreement. South Africa ammended the Environmental Conservation Act 73 of 1989 by developing plastic regulation in terms of section 24. This regulation came as a result of problems associated with the collection and disposal of plastic bags which resulted in pollution and degradation. The problem was mainly affecting low income areas where refuse removal services are inadequate. The regulation's main aim is to restrict the production of non-reusable plastic bags, and unnecessary use of excessive amounts of disposable thin plastic film for packaging.

6. Materials and methods

6.1 Quantitative and qualitative method

The quantitative and qualitative methods were applied during the study. This incorporated questionnaires and interviews, field surveys and data presentation.

6.2 Quantitative method

This method was applied through weighing waste generated in all the different waste generators. It was applied through field surveys that were conducted for data collection from households and analysed to address the research objectives.

6.3 Qualitative method

Structured questionnaire was used as one of the data collection methods.. This questionnaire was used to collect information from the municipality officials through an interview regarding waste management services and practices for Polokwane city. The questionnaire was structured for open- ended questions, where the municipality officials provided answers from questions that were asked, and close-ended questions, where some response and answers were provided.

6.4 Field survey

On-site waste separation and measurements were done at individual households from the three income groups at Ivypark, Florapark and Sterpark residential areas, representing low, middle and high incomes respectively. The three categories were based on the municipality categories of income which is done according to the size of the residential stand (Table 4). A 10 *l* plastic bin and 100 kg weighing scale were used to collect and weigh the wastes selected for sampling from households. Gloves and refuse bags were used for sorting the wastes; while facemasks and worksuits were used for protection during the sampling and measurement period.

Income level	Size of the residential site
Low	0 - 300 m^2
Medium	300 – 500 m^2
High	500 m^2 +

Source: Polokwane Spatial Development Framework

Table 4. Classification of low, medium and high level incomes based on the size of the residential space occupied

The formula below was used to determine the number of samples in all the three income groups:

$$Wg = (W_t - W_b) \tag{1}$$

Where:
Wg **=** Waste generated per income group per week,
W_t = gross weight of bin and waste
W_b = weight of empty bin
First the weight (W_b) of empty bin, using the weighing balance, was determined. Thereafter, the bin was filled with the sorted waste, while shaking the bin constantly to fill the voids. The difference corresponded to the weight of the waste.

6.5 Data analysis

The data obtained were subjected to statistical analysis in order to establish whether there was any significant relationship between the quantity of waste obtained and the income groups. The significant relationship was based on 95% level of confidence. The proportional allocation of samples in the three income groups was based on the formula used for stratified sampling which was as follows:
Low income group

$$ni = \frac{Nix\ n}{N} \tag{2}$$

Where: ni = sample size per income group,
Ni =Total population per income group,
N= Total population of the three income groups
n=General sample size of all the three income group
A total of 325 households were sampled out of 2111 households within the three income groups. The distribution of the sampled household was as follows:
Low income group (Ivypark): 77 households were sampled out of a total of 500 households
Middle income group (Florapark): 194 households were sampled out of a total of 126 Households.
High income group (Sterpark): 54 households were sampled out of a total of 350 households.
To calculate the total waste generated by each income group, the following formula was used:

$$W_a = \frac{W_b}{W_d} \tag{3}$$

where:
W_a = Total waste generated day/income group,
W_b = Total no of households sampled
W_d = No of days in a week
Similarly, to calculate the amount of waste that was generated per day per household involves the following formula:

$$W_k = \frac{W_a}{W_h} \tag{4}$$

Where:
W_k = Total waste generated /day/household,
W_a = Total waste generated/day/income group and
W_h = No of households sampled/income group

7. Results and interpretation

7.1 Waste generation

The study focused on the household solid waste generated within the three selected residential areas of Polokwane city, namely: Low income-Ivypark, Middle income-Florapark and High income-Sterpark (Table 5). Food waste was the highest across all the income groups with a percentage waste generation of 34% (Table 5a). The trend of wastes was as follows: Paper-20% > plastics-18% >glass-11% > cans- 11% >garden waste –6% (Table 5b). represent waste composition generated per household per day per person from the income groups. The mean composition of waste generation in the three groups is presented in Figure 3.

Waste component	Low -income group (kg/week)	Middle income group(kg/week)	High-income group (kg/week)	Total waste generated (kg)/week	Average waste generated (kg)/week
Paper	183	658	422	1263	421
Cans	153	437	88	678	226
Glass	261	347	112	719	240
Plastics	181	571	406	1158	386
Food wastes	341	1227	640	2208	736
Garden waste	194	154	33	381	127
Total waste generated per week	**1313**	**3392**	**1702**	**6406**	**2135**

Table 5. (a) Total waste composition from the three income groups

Waste component	Low - income group (%)	Middle- income group (%)	High-income group (%)	Average waste generated/week (%)
Paper	14	19	25	20
Cans	12	13	5	11
Glass	20	10	7	11
Plastics	14	17	24	18
Food wastes	25	36	37	34
Garden waste	15	5	2	6

Table 5. (b) Percentage of total waste composition generated per week from the three income groups

Waste component	Low -income group		Middle-income group (kg/week)		High-income group (kg/week)		Mean	
	Kg/household/day	Kg/household/day/person	Kg/household/day	Kg/household/day/person	Kg/household/day	Kg/household/day/person	Kg/household/day	Kg/household/day/person
Paper	0.39	0.05	0.48	0.08	1.11	0.18	0.66	0.11
Cans	0.28	0.04	0.32	0.05	0.23	0.03	0.27	0.04
Glass	0.14	0.02	0.26	0.04	0.29	0.04	0.23	0.03
Plastics	0.33	0.05	0.42	0.07	1.07	0.17	0.60	0.10
Food wastes	0.63	0.10	0.90	0.15	1.69	0.28	1.07	0.17
Garden waste	0.36	0.06	0.11	0.01	0.08	0.01	0.18	0.03
Total waste generated	**2.1**	**0.32**	**2.5**	**0.4**	**4.47**	**0.7**	**3.01**	**0.48**

Table 5. (c) Total waste composition generated per household per day per person from the three income groups

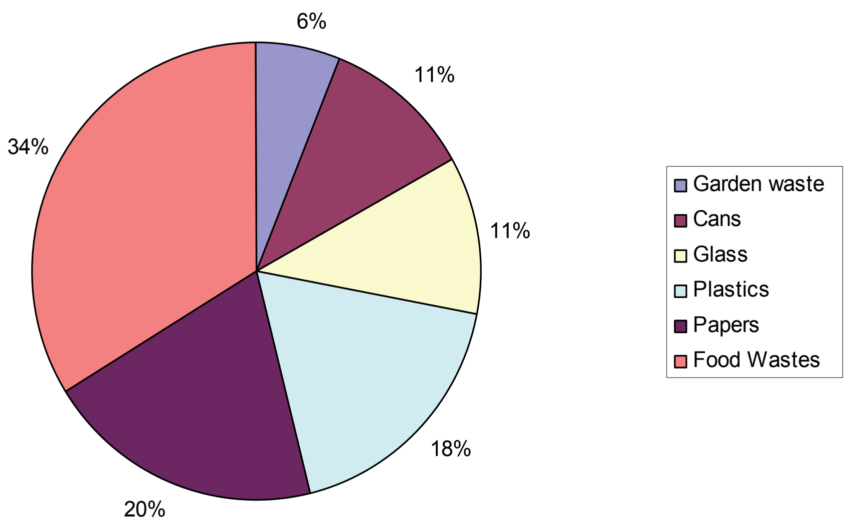

Fig. 3. Mean composition of waste generation for the three income groups

7.1.1 Waste generation in the low income group

The waste generated income in the low group was as follows: food waste 25% (341 kg/week) >glass -20% (261 kg/week) > garden waste – 15% (194 kg/week) > paper and plastic - 14% (181 kg/week plastic and 183 kg/week) > cans 12 % (153 kg/week) (Fig. 4).

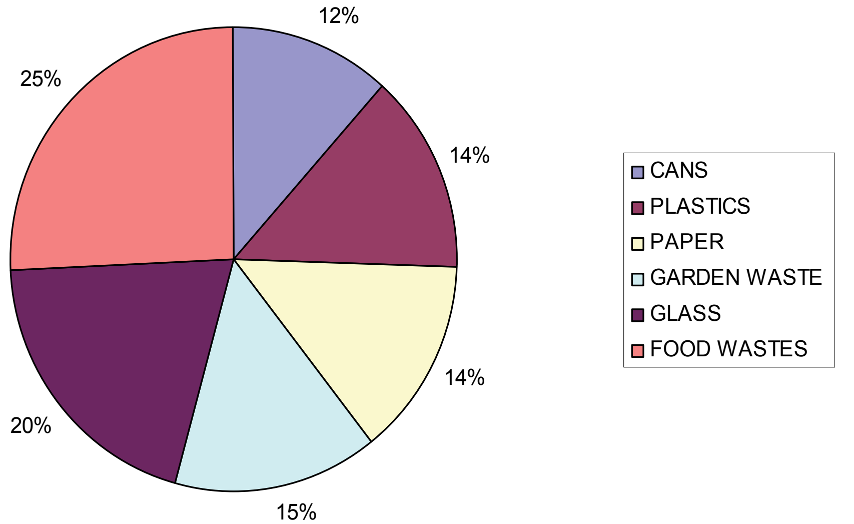

Fig. 4. Composition and percentage of waste generation from low income group.

7.1.2 Waste generation in the middle income group

Similarly in the Middle Income Group, waste was generated as follows (Fig. 5): food waste - 36% (1,226.50 kg/per week) > paper -19% (658 kg/week) > plastics - 17% (570.50 kg/week) > cans - 13% (436.50 kg/week) > glass - 10% (346.50 kg/week) > garden waste - 5% (153 kg/week).

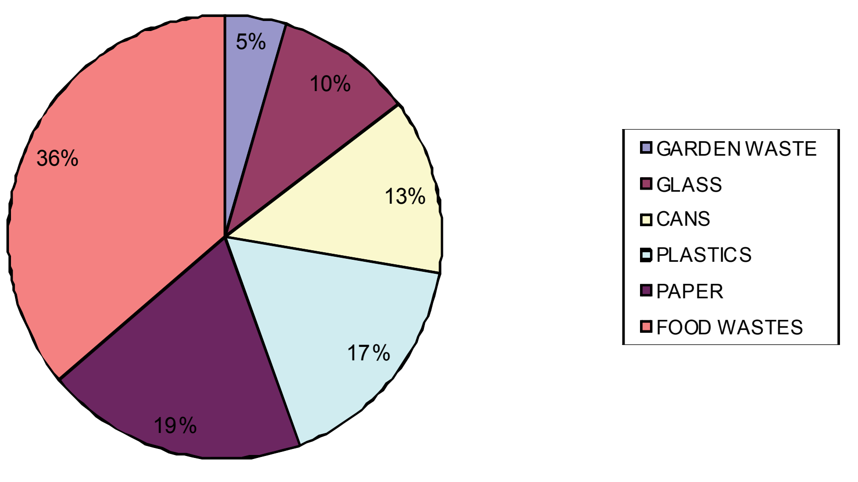

Fig. 5. Composition and percentage of waste generation from middle income group.

7.1.3 Waste generation in high income group

The composition and amount of waste generated was as follows (Fig. 6): food waste - 37% (640 kg/per week)> paper - 25% (422 kg/week) > plastic - 24% (406 kg/week) > glass - 7% (112.40 kg/week)> cans - 5 % (88.50 kg/week)> garden waste - 5% (33 kg/week).

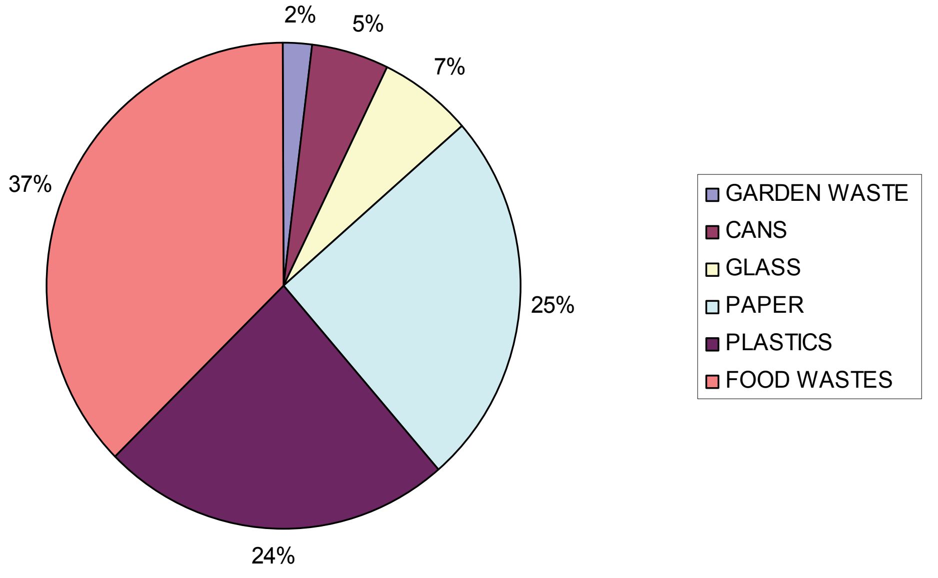

Fig. 6. Composition and percentage of waste generation from high income group.

7.2 Waste management system

Observations of the waste management system was made during sampling and follow up interviews were conducted with the personnel of the Department of Waste Management in Polokwane city, focusing on the waste management system, policies, municipality by-laws and regulations in place for controlling household waste. The response focused on Waste Management Policy, waste collection and transportation, and allocation of resources for refuse collection.

7.2.1 Waste management policy

Polokwane city is currently reviewing the refuse (solid waste) and sanitary by-law, the Administrative Notice No 845 of 1983, in line with the Integrated Waste Management Plan for the city. This notice addresses illegal dumping and sanitation related problems and penalties thereof in open places within the residential areas.

The policy has to be in line with the Constitution of South Africa 108 of 1996 and the Environmental Management Legislation, namely, National Environmental Management Act (1998), the Local Government Structures Act 117 of 1998 and the Local Government Municipal System Act 32 of 2000 which outlined the roles, responsibilities and the operations of all the municipalities. The development of Waste Management Plan is also in progress in order to align its function with the National Waste Management Strategy (1998) and Polokwane Declaration of Zero waste (2000).

7.2.2 Solid waste collection and disposal

It was noted that wastes from the households were not sorted. Instead, all the wastes collected from individual households were mixed in refuse bags. This makes recycling of wastes from homes not practical, and thereby reducing the quality of recyclable wastes like paper and cardboard through mixing of waste.

The waste refuse bags from households are collected weekly on a specific day for each suburb. For example, for Ivypark, collection is on Thursday, Florapark collection on Wednesday and Sterpark on Tuesday. The amount of waste collected on a weekly basis from the residential areas and city center amounts to 456 m^3. The collection system is quite effective, thus no refuse bag is left by the road side to litter the city.

There are four cooperatives involved in litter picking in the city with a total number of 47 workers and four ton truck for collection of waste from litter picking group. The municipality has allocated a total of 13 contractors that collect waste from residential areas in refuse bags and bins in the business area, 3 loadlaggers that collect solid waste from the skips in the factories, 7 grab that collect waste in transfer station and illegal dumping areas, and 3 multilifts for waste bins in the factories.

Waste was being disposed in one permitted waste disposal site, named the Weltevreden Landfill. The permit was issued in 1998 by the Department of Water Affairs and Forestry in terms of the Environment Conservations Act of 1973. In this case, Polokwane landfill had a license for operations which most municipalities in the Limpopo province do not have. Johansen and Boyer (1999) indicated in their study that South Africa is the only country in Africa with specific regulations and guidelines in place governing solid waste landfills. The minimum guidelines requirements for landfill classify land fills in terms of type of waste, size of waste stream and climatic conditions with focus on leachate generation. Polokwane landfill has been licensed as a G: M: B site which allows disposal of dry industrial waste, dry domestic waste, builder's rubble and garden waste. This classification allows for disposal of General waste, size is Medium, B- climatic water balance with no leachate management system required based on site specific factors of rainfall and evaporation rate (DWAF, 1998).

7.2.3 Waste recycling

Currently, there is no recycling programme implemented by the Municipality of Polokwane City. It has been found that 60% of waste disposed in the landfill consists of recyclable waste. Although the Municipality does not have a formal waste recycling system, it was found that the disposal site has informal waste reclaimers that are collecting recyclable wastes on a daily basis. This has also led to the development of an informal settlement close to the landfill. Waste reclaimers collect all the waste that is re-usable/recyclable ranging from bricks, plastics, steel, card boxes and cans (Fig. 7). Interview was conducted with the waste reclaimers to get data on the amount of recyclable waste collected per day. Unfortunately they never kept records of the amount collected apart from the price per Kilogram. For example, plastic- 60 cents/kg, aluminum cans-R 2/kg, cardboxes-R30/kg, plastic 2l cold drink containers -80 cents/kg, plastic milk containers -50 cents/kg, copper-R 15/kg brass R 4/kg. They were able to quantify the amount of money received at the end of the month which was approximately R300 per person, depending on the rate of collection for every individual.

Consultation with the recycling agent that collects waste from the reclaimers indicated that a total of 2,7120 kg recyclable waste was being collected from the landfill site daily, then sent to large recycling industries in Gauteng for further processing.

Fig. 7. Waste recycling by local waste reclaimers at the landfill site in Polokwane city.

A total of 28,000 m^3 was disposed per month which comes to a total of 336,000 m^3 of waste disposed per year. The entrance of the landfill had a weigh-bridge (Fig 8), to weigh all the trucks disposing waste after collection.

Fig. 8. Weighing bridge at the entrance of the Polokwane landfill site.

7.3 Allocation of resources

Resources allocated for refuse collection are as follows: 12 workers and three drivers with three trucks that are used to collect waste within the residential areas. Each Labourer is given a set of protective clothing comprising 4 overalls, 2 pairs of boots, 1 pair of rain coat per year and 1 pair of gloves monthly.

Currently, the Polokwane Municipality makes provisions of about R38,000,000.00 for refuse removal for the whole municipality. This budget is also supplemented by the monthly refuse removal services fee paid by residents. The fee is calculated based on the Local Government Municipal Property Rates Act No 6 of 2004 of South Africa (Table 6). These fees are revised annually based on the inflation rate and on the size of the size of the standirrespective of the income levels of different residential areas. The municipality issues out a utility bill on a monthly basis which incorporates the assessment rates for the property, sanitation, refuse removal, electricity and water.

Size of residential site	**Rates payable by residents refuse removal & sanitation effective from 01/07/2008**
0 - 500 m^2	R20.05
500 – 1000m^2	R52.25
1000m^2 +	R93.85

Source: Polokwane Municipality rates policy (2008)

Table 6. Rates for sanitation and refuse removal for Polokwane City

8. Discussion and recommendations

8.1 Waste generation

Globally, the rate of waste generation has increased over the years in different societies with increase in population and change in lifestyle and technological advancement. Recent research results reflect a population increase of 8.2 % from 2001 to 2007 in South Africa, and 9.5 % in Polokwane (Statistics S.A-Census 2007). This means that more waste is being disposed of into the landfill. As the number of land space for the establishment of landfill sites is becoming limited, other methods of waste management should be sought. This is where recycling programmes are expected to play a vital role in prolonging the life-span of landfill sites.

Waste generation in the three income groups was computed to be 0.3-0.7 kg per person per day, which was distributed as follows: low income group at 0.3 kg per person; middle income group at 0.4 kg per person and high income group at 0.7 kg per person. This amount of waste generated was low as compared to the findings of the Baseline Studies (DWAF, 1998). where the average amount of waste generated per person was found to be 0,7 kg per person in South Africa. Generally, it was observed that the amount of waste generated by the three income groups depended on the socio-economic level of the group. The High

income group was found to generate more waste than the low and middle income groups. This was attributed to the affordability of goods by this income group.
It is worth noting that the waste generated per person in Polokwane city is lower than that generated per person in Johannesburg. For example, in Johannesburg, the average waste generated per income group ranged from 0.4-0.7 kg per person, 0.7-1.1 kg per person and 1.2-2.5 kg per person for low, middle and high income groups respectively (City of Johannesburg, SOER, 2003). This is rather not surprising since most of the people residing in Johannesburg earn more than their counterparts in Polokwane and, therefore are expected to afford more goods which are disposed of after utilization.

8.2 Waste composition

Food waste constitutes the highest percentage of waste generated in all the income groups, although the percentage varied with the high income group having 37%, middle group-36% and low income group-25%. The waste composition found in the three income groups varied markedly. While the waste from the low income group had the highest percentage of grass waste, and that from the middle and high income groups were composed mainly of recyclable waste: plastics, glass, paper and cans.
Studies conducted in Nairobi agree with the data in Polokwane city that household waste comprised high percentage of food waste in all the three income groups sampled. Almost 50 % of waste generated in Nairobi was food waste (Henry et al., 2006), whereas in Polokwane food waste comprised 34 % of the total waste generated from the income groups. The studies in Nairobi also stated that 50 % of waste disposed of in landfills in that country is mostly organic waste which can be reduced by composting before disposing into landfills.
The results of this study shows that in Polokwane the amount of organic waste generated amounts to 40% which is low as compared to other studies conducted in Nepal, where organic wastes was 70% of the total waste generated, and 60 % recyclables for Polokwane versus the 20.5% (comprised recyclable waste such as cans, plastics and papers generated) (Viraraghan and Pokkhrel, 2005). According to the studies carried out in Macao in China, food waste accounted for 15 % of the total waste generated, and 52% was of recyclable waste (Jin et al., 2005).

8.3 Waste recycling

This study indicated, that about 60 % of wastes generated can be recycled. This included glass - 11%, plastics -18 %, paper- 20 % and cans-11%. The amount of potentially recyclable waste in Polokwane city is much high as compared to other cities for example, Nairobi 35 %, Macao-China 52 %, Singapore 30 % and Kathmandu 20.5 %, (Bai and Suntato, 2002). Although the Municipality does not have a formal waste recycling system, it was found that the disposal site had informal waste reclaimers that are collecting recyclable waste on a daily basis. This has also led to development of an informal settlement close to the landfill. Waste reclaimers collect all the waste that is re-usable, ranging from bricks, plastics, steel, card boxes, cans.
No informal recycling programme exists in Polokwane Municipality whereas other Municipalities such as the City of Johannesburg and the City of Cape Town have initiated recycling programmes. This is one area that the Municipality must explore in order to

achieve the Polokwane Declaration target on Zero Waste. Trois et al. (2008) investigated the idea of zero waste in emerging countries using South African experience. In this study, two communities, adjacent to the Mariannhill Landfill site in Durban were selected as a case study for comparative analysis of formal and informal settlements. On the basis of the results of the analysis of the recyclable yields and information provided by the questionnaire, a waste minimization scheme was proposed for zero waste schemes. This scheme lays responsibility on households to recycle their waste at source. It makes use of existing recycling strategies currently applied in other urban areas in South Africa such as drop off, kerb-side and central sorting. In another study by the same authors (Trois and Simelane, 2010), studied the possible implementation of separate waste collection and mechanical biological waste treatment. This model advocates pretreatment of the waste in passively aerated open windows for 8 weeks before finally taking it to the landfill. The pretreatment leads to volume reduction due to reduced biodegradable compounds in the municipal solid waste.

8.4 Waste collection and transportation

Domestic waste is collected from households weekly by the Municipality trucks. The Municipality has sub-contracted litter picking co-operatives to pick up litter along the streets in the residential areas and finally dispose at the landfill. Litter collected is not sorted into recyclables or non-recyclables; is all disposed to the landfill with no sorting, which could be another area where the Municipality can initiate a recycling programme through the litter picking cooperatives.

It has been outlined in the baseline studies on waste generation conducted in 1998 and State of Environmental Report (2003) for City of Johannesburg that over 50% of waste going to the landfills has the potential to be recovered for recycling or re-use. Based on the information from the Municipality, a total of 28,000 m^3 of domestic waste is disposed of in Polokwane landfill. Out of this 60% of waste generated in the households can be recycled, if proper waste recycling system is put into place.

9. Conclusion

The current study established the following:

- The level of income of each household group determines the volume of waste generated by such a group, thus the higher the level of income for the group, the more waste it generates.
- It was observed that volumes of waste and composition were not the same in each household group but this depended on the lifestyle, for example, the high income group had the lowest garden waste since they can afford private garden services that dispose garden waste after its generated, as compared to the low income group. The high income group also had the highest percentage of paper waste in a print form, which is linked to affordability.
- The amount of waste from households in Polokwane Municipality that can be recycled constituted about 60%. This could significantly reduce the amount of waste being disposed into landfills. The implications of this strategy would lead to the achievement of the Polokwane Zero waste declaration target of 50% waste reduction by 2010 and zero waste generation by 2020.

10. Recommendations

- There is a need to develop an integrated waste management plan for Polokwane city with a priority on waste recycling to reduce the final amount of waste for disposal.
- The Polokwane Municipality should develop an environmental awareness programme on recycling supported by placement of the recycling containers at strategic points to collect recyclable waste.
- The Municipality should come up with a strategy of supporting household separation at source.
- The programme of cooperatives for litter picking should be extended to include recycling.
- Waste generation is a vital component of waste statistics. The accuracy of these statistics is important in planning, development and monitoring waste management strategies. The Municipality should develop a monitoring system for waste classification, quality and quantity.

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Part 2

Processing of Solid Waste

7

Dry Digestion of Organic Residues

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1. Introduction

Sustainable development closely links to the context of energy. Replacing fossil fuels with sustainably produced biomass or organic residues will not only be a way to cope with the depletion of fossil fuel resources but also to reduce the CO_2 emissions into the atmosphere and therefore minimise the risk of global warming. A large variety of methods of biomass energy conversion are available today. Some technologies produce secondary fuel such as methanol or biomass oil which then can be utilized for various purposes. Especially for electricity generation efficient processes might be direct combustion, thermal gasification or the production of biogas.

Anaerobic digestion (AD) with biogas production, including utilisation of the organic fraction of waste materials and of residues, is a particularly promising choice and experiences increasing interest worldwide. AD does not only supply a clean and versatile energy carrier, thus displacing other energy sources such as fossil energy, but is well suited to contribute towards appropriate waste management schemes in urban areas and in agriculture. Biogas production has high potential worldwide, and it is in particular digestion of solid materials which is of increasing interest. As a result, it is to be expected that so-called dry digestion systems, operated with an elevated content of total solids (TS) in the reactor, will experience more widespread implementation.

Agricultural residues in general are left on field or are brought back to field in order to supply fertilizers and to improve soil quality. Anaerobic digestion offers the possibility to produce renewable energy, and at the same time generates a digestate with an improved fertilizer value.

Currently the most common strategy for management of municipal solid waste (MSW) worldwide is still landfill. As a result of higher environmental awareness, and often based on favourable legislative backgrounds, more and more emphasis is given to recycling and recovery, and in particular to efficient use of organic materials. Composting and anaerobic digestion are state-of-the-art for treating organic substrates.

Germany today is leading in the area of biogas production. Around 6,000 AD plants are in operation, and the number is further increasing. Most plants are in the agricultural sector, but currently at least 100 plants are run solely on the organic fraction of MSW, a direct result of introduction of source-segregation schemes for household waste. Due to favourable frameworks the number of AD plants in the municipal field will significantly rise in the coming years (along with more agricultural plants to be build).

2. Anaerobic digestion: basics

Different types of biomass can be used for biogas production, including organic waste from gastronomy/food waste, the organic fraction of MSW, organic waste from industry/commercial waste, sewage sludge, excreta, agricultural residues, and for energy generation purposes grown energy crops. This book chapter focuses on biogas production with solid waste materials. The main principles are common in digestion of all materials, though solid substrates require adaptation of the processes. Separate collection of organic fractions and diversion from landfill is among the main success criteria.

2.1 Principles and products of the AD process

Biogas is produced in the absence of oxygen (anaerobic digestion) through biological activity of different microorganisms if the environment is friendly for the microbes (water content, temperature, nutrients). The substrate must provide all components necessary for the metabolic processes (C, N, O, H, S, P, K, Ca, Mg), including micronutrients such as nickel, iron, zinc, manganese, copper, molybdenum, selenium, wolfram. Material should not have inhibiting substances (e.g. disinfectants, antibiotics, heavy metals). Inhibitory or toxic effects are in general related to concentration and process conditions. As metabolic (intermediate) products can also have inhibitory effects (NH_3, H_2S, volatile fatty acids, H_2), process conditions need to be controlled.

Anaerobic digestion with biogas production is the result of an anaerobic reaction chain with several steps. Each of the steps hydrolysis, acidification, acetogenesis, methanogenesis involves specific groups of microorganisms with individual requirements. Efficient biogas production necessitates that process conditions are favourable (or at least tolerable) for each of the groups. Microbiology, together with different characteristics of manifold potential substrates, is one explanation for the large variety of technical solutions to be found in full-scale applications.

During anaerobic digestion a large part of the energy contained in the biomass is transformed into methane, an energy carrier which then can be used for example to produce electricity. Anaerobic digestion of glucose for example leads to biogas which contains 85 % of the energy content of glucose (2868 kJ/mol contained in glucose after having been formed in the photosynthesis pathway), see Fig. 1.

Biogas has a wide variety of possible applications, the most common ones are:

- Direct use for cooking and lighting (small-scale AD plants at household level)
- Utilisation for heat generation
- Generation of electricity (several engine types can be fuelled with biogas; electricity generation is often accompanied by heat generation in combined heat and power plants/ CHP)
- Fuel for cars/vehicles
- Feeding into the natural gas grid (after upgrading to natural gas quality; now one standard in industrialized countries when produced at large scale; different upgrading technologies exist)

Compared to other renewable energies, it is one advantage of the energy carrier biogas that it can be stored to be used according to fluctuating demands or to availability of alternative energies. Biogas can be a particularly advantageous choice e.g. in hybrid power systems for electricity supply in remote areas or islands (Borges Neto at al., 2010). It is not necessary to make use of biogas directly at the production site. Local biogas grids can be an intelligent solution to provide biogas to where it can be used at highest efficiency (Panic et al., 2011).

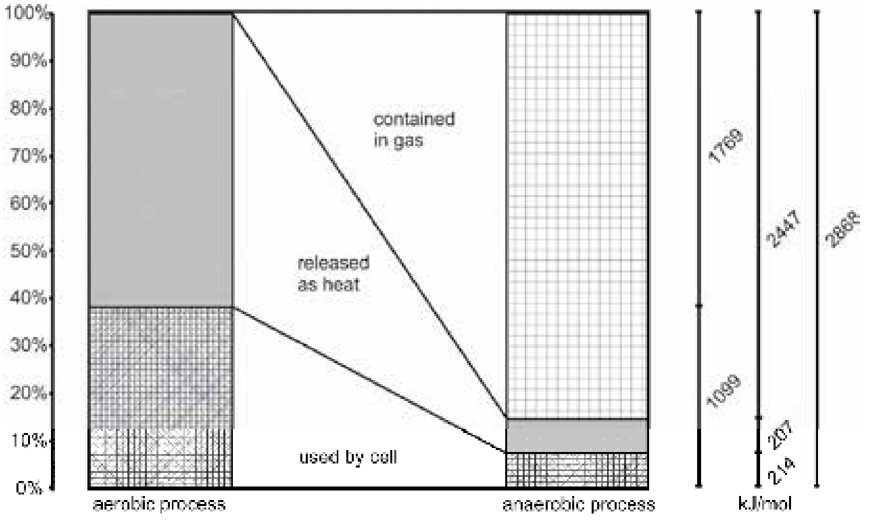

Fig. 1. Energy balance of aerobic and anaerobic degradation of glucose (based on Kranert, 1989)

During digestion, the amount of organic material is reduced in the substrate whereas nutrients like nitrogen are conserved in the biomass. AD residue therefore is an efficient fertilizer. Especially with regard to nitrogen biogas residues have excellent nutritional properties as digestion encourages transformation into bioavailable ammonia. The extent of nutrient uptake by plants depends on the time of application and there is always the possibility that nutrients will be leached from the soil when plants are unable to take them up. While in the organic form nitrogen must be first mineralised, AD converts much of the organic N into ammonia, yielding a digestate with 60-80% of the total nitrogen content in the form of ammonia (Banks et al., 2007). This makes it highly predictable, minimises leaching losses and is in line with the development of good agricultural practices. Ammonia can be converted to nitrate for plant uptake, while some plants may use ammonia directly.
The improved fertilizer value of AD digestate is to be considered as economic advantage of the AD unit. Other fertilizers are displaced and higher biomass yields are possible, as has been reported for napa cabbage, cauliflower (Jian, 2009). Digestate which is not fit for landspreading (e.g. due to contamination with heavy metals) must be disposed of.

2.2 AD plant types

Many different AD plant types have been developed and are to be found in full-scale for various applications and in different regions. The following overview is restricted on types typically implemented for digestion of solid waste materials, agricultural substrates and household wastes. Table 1 provides an overview on different technology concepts.

Operation of mode: batch, fed-batch or continuous	In *batch systems* the whole substrate is filled at once into the reactor and is digested over a pre-defined period. When digestion is complete material is removed and the process is started with a fresh load. In batch systems digestion and methane production start anew with each filling of the reactor and biogas supply therefore is not continuous. For commercial operation it is in general necessary to have several reactors run off-set (alternative loading and unloading), at least three reactors should be operated.
	In *fed-batch mode* material is added to the digester by and by until the space is used up. Then all material is removed and the emptied digester provides new reactor volume.
	In a *continuous* system (or more precise semi-continuous) substrate is regularly fed into the reactor, and at the same time effluent is unloaded. Biogas production is continuous. Such a system in most cases is judged to be better suited for large-scale operations (Suryawanshi et al., 2010), drastic changes of input composition should be avoided.
Transport of material, homogeni-sation in reactor	The most common types of AD plants are based on the concept *continuously stirred tank reactor* (CSTR). Plants are equipped with facilities for stirring the digester content (continuously, or in most cases semi-continuously), resulting in homogenization of reactor content but also in differing retention times for different particles, with part of the material leaving the reactor after very short digestion.
	Plug flow digesters are long narrow reactors (typically 5 times as long as the width) with inlet and outlet at opposite ends. Feeding is carried out semi-continuously and typically with a thick substrate (~15% TS). In general there is no internal stirring device, material advances whenever new substrate is added and in theory the reactor content does not mix longitudinally on its way towards the outlet (but actually material does not remain as a plug and portions advance faster than others – but minimum retention time is assured far better than in CSTR concepts, thus allowing for better hygienisation).
Total solids content (TS)	So-called *wet digestion plants* are most common in agriculture, they are operated at TS < 12%. When digesting higher amounts of solid materials, water content needs to be adjusted (addition of liquid substrates, water or recirculation of digester effluent).
	For digestion of organic materials available mainly in solid form, implementation of technical processes designed for higher TS contents was a logical step (e.g. municipal bio waste). So-called *dry digestion plants* are typically operated at TS > 20%, water content often is not adjusted to a specific value but is a result of the digesting substrates.
	It needs to be mentioned that no final definition based on TS content exists; in literature other TS limits can be found. Occasionally a third type is introduced in order to characterise processes operated between 12 to 20% TS: *semi-dry digestion.*
Digestion temperature	Most AD plants are operated in the *mesophilic range*, optimally around 30-38 °C. Especially in tropical countries AD plants are operated without

	temperature control with digestion at ambient temperatures (~20-45 °C). Mesophilic processes are more stable than thermophilic, the greater number of mesophile microorganisms makes the process more tolerant to changes in environmental conditions.
	Besides mesophilic AD, *thermophilic digestion* is a conventional operational temperature, optimally around 48-57 °C. The increased temperature results not only in better hygienisation, but in faster reaction rates, and consequently faster biogas production (shorter retention times, higher degradation rates). However, the process is less stable and requires higher energy input for reactor heating.
	AD plants operated at *psychrophilic temperature* (<20 °C) are less common, they are restricted to low-tech applications. Degradation of organic material and biogas production are very slow, resulting in long retention times.
One-, or two-stage (multi-stage) systems	In *two-stage systems* (or multi-stage systems, which however are very rare) process conditions can be optimized for the different groups of microorganisms in order to improve overall efficiency. While during the first phase conditions can be optimized in order to achieve a rapid liquefaction, the second phase converts soluble matter into biogas. Compared to single-stage systems the process is more rapid and more stable, but investment and maintenance costs are considerably higher.
	By far the most AD plants are *one-stage processes*, with one single reactor for the digestion process (in general followed by a storage tank).

Table 1. Types of digesters

2.3 Digestion at elevated TS contents

In agriculture, continuously operated reactors processing materials with high water contents are most common. This is to be explained by the fact that slurry was the predominant substrate for agricultural biogas plants throughout many decades. However, the use of solid substrates such as yard manure and especially energy crops is becoming more attractive (Amon et al., 2007; Weiland, 2006). In full scale, digestion of solid biomass is limited in conventional slurry-plants, due to technical restrictions in particular related to mixing and feeding devices, and technologies appropriate for operation with elevated content of total solids (TS) are imperative.

In general, the term 'dry fermentation' describes digestion with higher TS content. Since lack of moisture limits bacteriological activity in all anaerobic systems, no digestion can actually be 'dry'. Therefore, the terms 'solid-state' digestion (Martin, 1999; Martin et al., 2003) or 'solid-phase' digestion (Anand et al., 1991; Chanakya et al., 1997; Kusch et al., 2008) are used as equivalents to 'dry digestion'. Similarly, the term 'liquid-phase digestion' is used as equivalent to 'wet digestion' for processes operated with low TS content.

Dry digestion is still uncommon in agriculture, but to treat municipal solid waste so-called dry digestion processes with > 20% TS are implemented to at least a similar extent than wet digestion processes (Bolzonella et al., 2003; Forster-Carneiro et al., 2008), in general one-stage processes are favoured (Forster-Carneiro et al., 2008). As MSW is a solid material, development of technological concepts adapted to high TS contents was a logical step. While MSW is mainly processed continuously, batch processing of solid material prevails in agricultural dry digestion systems.

Both, single-stage (Kusch et al., 2008; Svensson et al., 2006) and two-stage approaches (Andersson & Björnsson, 2002; Linke et al., 2006; Parawira et al., 2002) are the subject of research for both, batch and continuous processing. Section 3 of this Chapter describes in detail a full-scale application and experimental results of one-stage batch dry digestion, and Section 4 focuses on two-stage continuous processing.

Dry digestion reduces the risk that process problems will occur due to fibrous materials floating on top of the liquid, a phenomenon often observed in wet digestion of lignocellulosic substrates such as straw or straw-containing dung, e.g. from horses (Kalia & Singh, 1998). Some further advantages associated with dry digestion systems are as follows (Hoffmann, 2001; Köttner, 2002): lower reactor volume, less process energy, lower transport capacity, less water consumption.

Monitoring results of 61 full-scale AD plants revealed that in particular continuously operated dry digestion plants are comparable to wet digestion in terms of general efficiency, methane productivity (methane generation per net reactor volume and day) and methane yield (FNR, 2009). It was however pointed out that demands on technical equipment (stirring devices, pumps) are much higher, due to the elevated viscosity of the digester content. In addition, there is a higher risk for shortage of micronutrients, and as a result addition of micronutrients is common. Discontinuous dry digestion in box type digesters was found to have a higher risk for lower gas yields and for increased odour emissions due to handling of material outside the digestion boxes (see Chapter 3), but the monitoring project confirmed that plants are robust and failure rarely occurs.

2.4 Degradability of solid residues

Lignocelluloses comprise a large fraction of solid biomass such as MSW, crop residues, animal manures, woodlot arisings, forest residues or dedicated energy crops (Sims, 2003). Global crop residues alone were estimated at about 4 billion Mg for all crops and 3 billion Mg per annum for lignocellulosic residues of cereals (Lal, 2009). Biogas production from lignocellulosic biomass is a slow (without pre-treatment having been applied to the substrate prior to digestion) but steady process. Methane originates mostly from hemicellulose and cellulose, but not from lignin which cannot be degraded by anaerobic microorganisms. As in other biochemical conversion pathways, in the anaerobic digestion of this substrate type, enzymes must first break the lignin barrier in order to gain access to the degradable components. In order to make these biomasses better available to anaerobic degradation, various physical or chemical pre-treatment technologies are known, including thermochemical or ultrasonic pre-treatment, use of different additives or steam pressure disruption (Liu et al., 2002; Petersson et al., 2007; Yadvika et al., 2004). Though potentially applicable on larger scale, for lignocellulosic materials contained within solid manure sophisticated expensive pre-treatment procedures seem inappropriate for utilisation on single farms. Two-stage digestion with hydrolysis is feasible (see Chapter 4.2).

The actual methane yield depends on the total methane potential, the digestion time and degradation kinetics (influenced by substrate characteristics and process conditions). The total methane potential $G_{pot} = G(t \to \infty)$ can be determined by optimized batch testing, which should include extrapolation of the experimental findings (Kusch et al., 2008; 2011). The exploitation degree $q_t = G_t/G_{pot}$ indicates the proportion of G_{pot} released at a specific point in time (t). Table 2 lists selected experimental results.

	q_{26}	q_{28}	q_{42}	q_{49}	q_{74}	
Oat husks	0.61	0.65	0.84	0.90		Kusch et al., 2011
Horse dung with straw		0.52	0.62		0.74	Kusch et al., 2008
Wheat straw	0.49		0.61			based on Møller et al., 2004

Table 2. Exploitation degree $q_t = G_t/G_{pot}$ for different digestion times

2.5 Legislative background for diversion of MSW fractions from landfill to AD

The organic fraction of MSW is most suitable for biogas production through AD, and as mentioned above dry digestion is particularly well suited and most common. Among the key factors towards more widespread implementation of AD for organic MSW fractions, is waste segregation at source. The existence of legal frameworks resulting in authorities being liable to promote source-segregation in order to avoid landfilling of biodegradable waste is one of the main drivers to dissemination of AD in a country. Since several decades, legislation in the area of treatment of MSW has placed increasing emphasis on recycling and recovery in Europe and in many other countries.

The effect of legislation can be shown on the example Germany. Today the country has one of the highest recycling rates in the European Union (EU) and worldwide, and a significant amount of energy is recuperated via combustion by waste to energy treatment facilities (WtE), with generation of electricity and heat. In 2009 ~67% of MSW was recycled, incineration was applied to ~32%, while only 0.4% of total MSW was landfilled (2 kg per capita, compared to 216 in 1997) (Fig. 2).

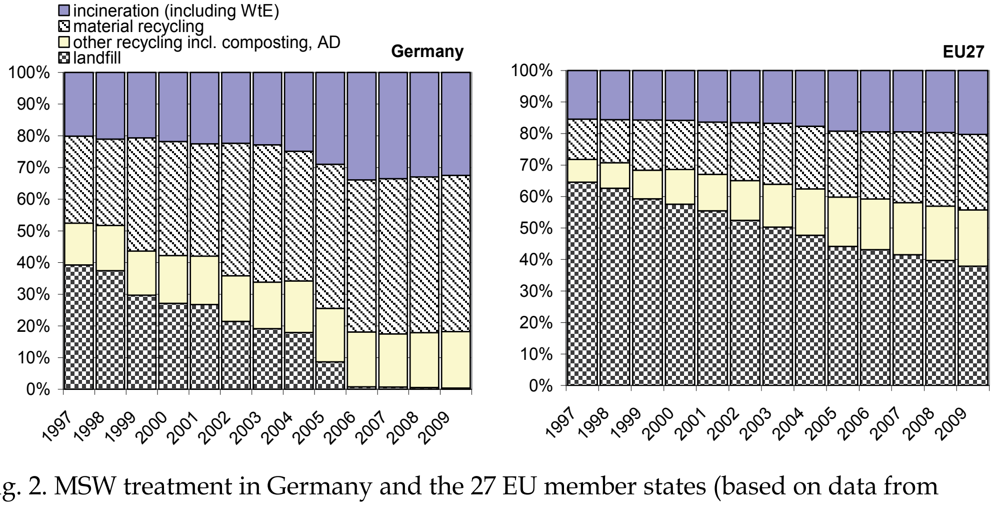

Fig. 2. MSW treatment in Germany and the 27 EU member states (based on data from Eurostat, 2011)

Germany is subject to EU regulations and has aligned its legislation according to demands on EU level, and often more stringent targets were set. Within the key focus of the EU Landfill Directive (1999) is to reduce negative effects of landfilled waste on the environment. According to the set EU objectives waste disposal is to be reduced by 20% by 2010 and by 50% by 2050 compared to 2000, and it is in particular the amount of biodegradable MSW

going to landfill which is gradually to be reduced (75% of biodegradable MSW going to landfill by 2006, 50% by 2009 and 35% by 2016 compared to a 1995 baseline).
Three legislation schemes have had highest impact on the high recycling rates in Germany (Mühle et al., 2010): (i) a refund system for cans and bottles ("Ordinance on the avoidance and recovery of packaging wastes", 1998), (ii) introduction of kerbside collection in the early 1990s (following the "Act for promoting closed substance cycle waste management and ensuring environmentally compatible waste disposal") and (iii) severe restriction of landfilling of non-pretreated MSW since 2005 by the commencement of the "Technical instructions on MSW" (replaced by the "Landfill Ordinance", which came into force in 2009). Landfill of non-pretreated MSW is now practically impossible, and it is in particular reduction of the organic fraction which needs to be ensured by pre-treatment.

3. Batch anaerobic dry digestion

Batch-wise digestion of stacked biomass represents a particularly simple system. More and more box type fermenters with percolation (sprinkling of process water over the stacked biomass) are to be found in full scale. The box type reactors process mainly agricultural solid substrates. Some more facilities digest municipal biowaste and have proven reliability (e.g. systems Bekon, Biocel).

3.1 Principles

Substrate is filled at once into the reactor and is digested over a pre-defined period. The addition of an appropriate ratio of solid inoculum accelerates methanisation and prevents digester failure (Ten Brummeler & Koster, 1989). The sprinkled liquid assures favourable biomass moisture content.
In order to equalise gas production at least three batch-operated dry digestion reactors need to be run offset. In general all digesters are functionally coupled through the recirculated liquid: leachate of all reactors is collected in a common process water tank and reused for percolation. It is not possible to operate the system without a separate process water tank, since the total volume of liquid varies in time and depends on water content, water holding capacity and degradation kinetics of the solid materials. Due to the water movement through the stack of solids, organic material is partly washed out from the substrate stack and is metabolised either in the liquid tank or in other solid-phase digesters, while only part of the total methane production actually occurs in the substrate itself.
Experimental results demonstrate that in batch-operated dry digestion with percolation significant amounts of biogas can originate from methanogenic activity in the process water tank. This gas volume is not to be neglected and represents a valuable energy source. If not valorised, the negative effect lies not only in the fact that the energy content is not utilised but also in the fact that any methane released to the atmosphere will function as greenhouse gas. There is a general tendency to keep this plant type as simple as possible. Experimental results suggest that gas capture not only from the digesters but also from the process water tank should be considered as a standard. Dimensioning of the process water tank is not of decisive influence on methane generation in the liquid phase. Even when deciding in favour of a small process water tank, equipment for catching generated biogas from the tank needs to be foreseen. Especially when digesting easily hydrolysable biomass, special attention needs to be given to biogas generated in the liquid phase (Kusch et al., 2009).

3.2 Description of a selected full-scale plant

Within a research project, full-scale experiments have been performed at a farm plant located in the southern area of Germany on a farm with organic farming (Bioland). The plant consists of four concrete digestion boxes of 130 m^3 each (Fig. 3). Process water was sprinkled over the biomass bed and leachate of all four boxes was collected in one tank to be reused for percolation. Digestion temperature was in the mesophilic range. Percolation (not automated) was around twice daily in routine plant operation. The full-scale farm plant has been described in previous publications (Kusch, 2007).
Though other materials (solid dung, grass, energy crops) were added as well, the AD plant was built especially for the digestion of green cut collected by the local authority. This material is not suited for conventional wet digestion due to the presence of stones and a high proportion of woody biomass. Green cut was chopped to <10 cm before digestion.

Fig. 3. Full-scale farm plant with four solid-phase digestion boxes

The fermenters were filled and emptied by using a wheel loader. Before the filling, substrate was mixed with solid inoculum (digested material from the previous cycle). For the mixing, a windrow was formed of fresh substrate and of solid inoculum and mixed with a compost windrow turner. A short period of pre-composting ensured that temperature of the substrate increased so that pre-heated material was brought into the reactor.

3.3 Experimental results

A combination of laboratory and farm scale experiments was conducted. The farm scale plant is described in Chapter 3.2. A dry digestion laboratory with 10 reactors (50 L each) was build (described in detail by Kusch (2007)).
Experiments showed that the necessary amount of inoculum strongly depends on specific substrate characteristics and may vary within a wide range (ensiled maize: around 70 % w/w based on TS; ensiled grass: around 70 % w/w TS; horse dung with straw: 10 to 20 % w/w TS; cattle dung: 0 %, but augmentation of gas yield in mixture with structure material). It was found that both in laboratory and full-scale achieved biogas yields were comparable to the yield obtained in liquid-phase digestion, if process conditions were optimal. However, suboptimal conditions resulted in an inhomogeneous and incomplete degradation at farm-scale (Kusch, 2007). Optimal conditions are difficult to be determined

and to be fulfilled at farm-scale, which increases the risk of incomplete degradation in this simple fermenter type with no substrate mixing.

It has been demonstrated that during process initiation discontinuous leachate recirculation is more favourable than continuous watering (Kusch et al., 2012; Martin, 1999; Vavilin et al., 2002, 2003; Veeken and Hamelers, 2000), which is assumed to be the result of encouraging methanogenic areas to expand throughout the whole digester, while continuous watering bears the risk to spread acidification. In addition, it was demonstrated that for the process type discussed here there is no beneficial effect of continuous water circulation compared to discontinuous watering throughout the whole digestion process (Kusch et al., 2012).

Experimental results (at laboratory scale) clearly indicate that methane formation within the recirculated liquid significantly adds to the total methane production of the process. In testing, up to 21% of the total methane generation originated from the recirculated liquid. This suggests that methane generation from the liquid phase is not to be neglected and needs to be recuperated. The process water tank should be equipped with gas collection facilities. Experimental results further indicate that higher ratios of easily hydrolysable substrates increase the proportion of methane from the liquid phase while slowly hydrolysable material encourages biogas generation in the decomposing biomass bed itself (Kusch et al., 2009).

The successful implementation of processes with percolation necessitates that liquid actually trickles through the whole substrate stack. Therefore, process water with low viscosity must be used as should substrate with sufficient structure. Liquid manure (slurry) is not suitable for percolation, as it will not ensure a leachate flow through the solid biomass bed. If no process water is available, fresh water (e.g. rain water) can be used to start the process.

Materials with poor structure should be mixed with structure material such as straw or green cut before digestion. In order to facilitate homogeneous digestion and avoid excessive tightening during the process, the fresh biomass stack should not exceed a height of 3 m.

Operation of the full-scale plant has proven to be robust and flexible, which are the main advantages associated with this process type. It needs to be taken into consideration that – in contrast to continuous process types – no process automation is possible, and as a result the necessary amount of effort and labour increases drastically for higher throughputs and higher numbers of reactors (the volume of one reactor is limited).

Choosing one process type among several alternative systems should depend on the specific characteristics of the available materials. If biogas generation is envisaged exclusively with energy crops, continuously operated process alternatives should be given special consideration. Discontinuous digestion with stacked biomass and sprinkling of process water is not the optimal choice for such substrates due to their poor structure and the high inoculum proportion required. Especially for materials such as energy crops with high costs for cultivation and conservation, incomplete degradation may have critical effects on the profitability of a biogas plant (a factor which is less relevant for digestion of organic residues). Therefore, compared to digestion of waste materials, special care should be taken so as to avoid inactive zones with inhibited degradation.

For discontinuous digestion with stacked biomass and percolation, structure-rich biomass, e.g. green cut or solid dung, is especially advantageous choice when considering process technology. Mixtures of structure-rich biomass and energy rich materials are well suited both in terms of material characteristics and energy production.

Successful implementation of discontinuous dry digestion is the result of two main factors:

- Favourable process conditions during digestion
- Appropriate choice of dry or wet digestion depending on the specific characteristics of the available substrates

4. Continuously operated dry digestion

Continuously operated reactors are commonly used in municipal waste digestion, they are state-of-the art. There are several manufacturers offering large scale plants.

Process	Waste	Capacity	TS in reactor	Retention	Biogas yield			CH_4 content
		Mg/year	%	days	Nm^3/Mg TS	Nm^3/Mg VS	Nm^3/Mg Input	%
3A	BW			45-50	410	285	100	55
BEKON	BW		≤50	28-35	240-530	170-370	60-130	55-60
KOMPOGAS	BW	20,000	35	15-20	380	245	85	50-63
ATF	BW	1,000	35-50	15-25	120-400	96-320	30-96	55-65
DRANCO	BW	20,000	18-26	20-30	550-780	390-550	120-170	50-65
DRANCO	OR	13,500	56	25	460-490	240-250	133-144	55
VALORGA	BW	52,000	30-35	24	390-410	175-185	80-85	55-60

BW: biowaste; OR: organic residues (municipal); TS: total solids; VS: volatile solids

Table 3. Technical parameters of large-scale organic waste disposal dry fermentation plants (as compiled by Kraft, 2004)

Though wet digestion prevails in the agricultural area, farm scale continuously operated dry digestion plants operated on organic residues are known as prototype or single farm specific solutions. The following focuses on innovative solutions for continuous dry digestion of agricultural organic residues.

4.1 Principles

One of the first prototypes for continuous dry digestion of agricultural substrates was developed in Switzerland (Baserga et al., 1994). It was a pilot plant of 9.6 m^3 capacity for continuous digestion of solid beef cattle manure on-farm. The solid manure was pressed via a pipe into the top of an upright standing cylinder. The pipe was heated to ensure that the material reaches the fermenter at suitable temperature. For discharging a scraper floor filled a discharge screw and the digesting residue was separated by a screw press. The liquid fraction was used as inoculum sprayed on the top of the reactor.

This principle was further developed by a prototype designed by Timo Heusala and implemented at Agrifood Research Finland (MTT) in Sotkamo, Finland (Virkkunen et al., 2010). Metener Ltd delivered the measuring equipment and modifications.The size of the screw stirred fermenter is 4.5 m^3 and the liquid volume is about 3 m^3, Fig. 4. A feeder screw charges solid manure from beef cattle. The manure is a mixture of excreta, peat, and straw or reed canary grass. The fermenter is discharged by a screw.

Also in Switzerland at FAT, a channel pilot reactor was developed. Baskets filled with solid manure pass through a slurry filled airtight fermentation channel. This solution did not find its way into praxis yet.

A continuously two stage two phase pilot plant was developed by Lars Evers in Järna, Sweden (Schäfer et al., 2005). This biogas plant is a suitable tool not only for renewable energy production but also for designing organic fertilisers by varying anaerobic process parameters like load rate of the reactor, retention time and mechanical treatment before, within and after the anaerobic process. This plant is described in the following chapter.

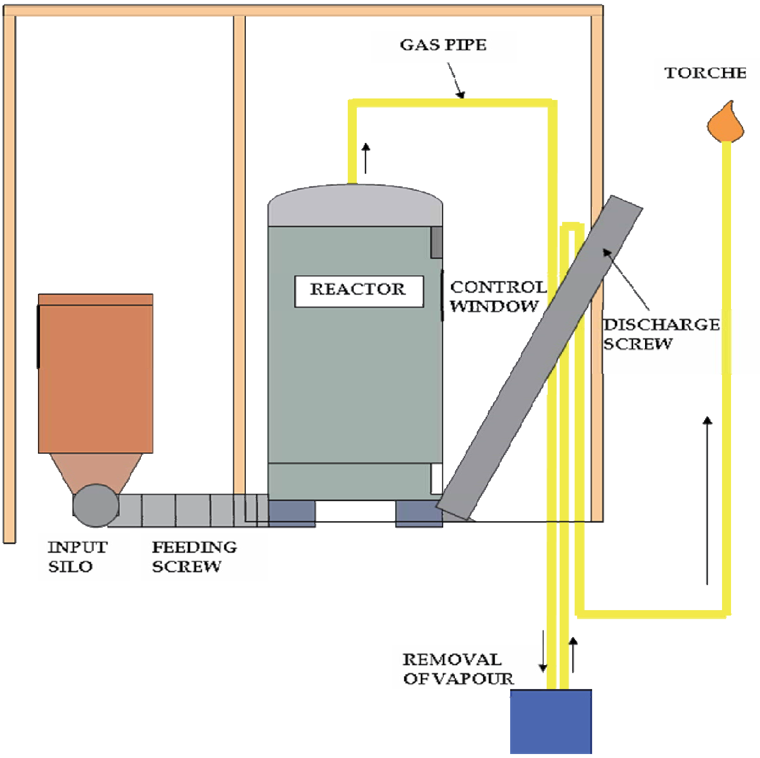

Fig. 4. Prototype of a solid manure fermenter in Sotkamo, Finland; picture courtesy Heidi Kumpula

4.2 Description of a selected full-scale plant

The local association of farms, horticulture enterprises, food processing units, food stores, and consumers in Järna aims to recycle organic waste. The goal is reduced use of non-renewable energy and use of the best-known ecological techniques in each part of the system, to reduce consumption of limited resources and minimize harmful emissions to the atmosphere, soil, and water. The biogas plant described here served as reference plant for nutrient recycling solutions within the BERAS-project of "The Baltic Sea Region INTERREG III B Neighbourhood Programme 2000-2006" of the European Union. Presently the biogas plant digests dairy cattle manure and organic residues originating from the farm and the surrounding food processing units.

The prototype plant is situated on farm close to the stall. Figure 5 shows the block diagram of the material flow of the two reactors. The blue boxes describe the processes, the white boxes the input and output, and the yellow boxes digestion residues within the process.

Both reactors are made of CORTEN-steel cylinders of 2.85 m inner diameter. They are coated by 20 cm pulp isolation and corrugated sheet. The steel cylinders were formerly used as smokestack.

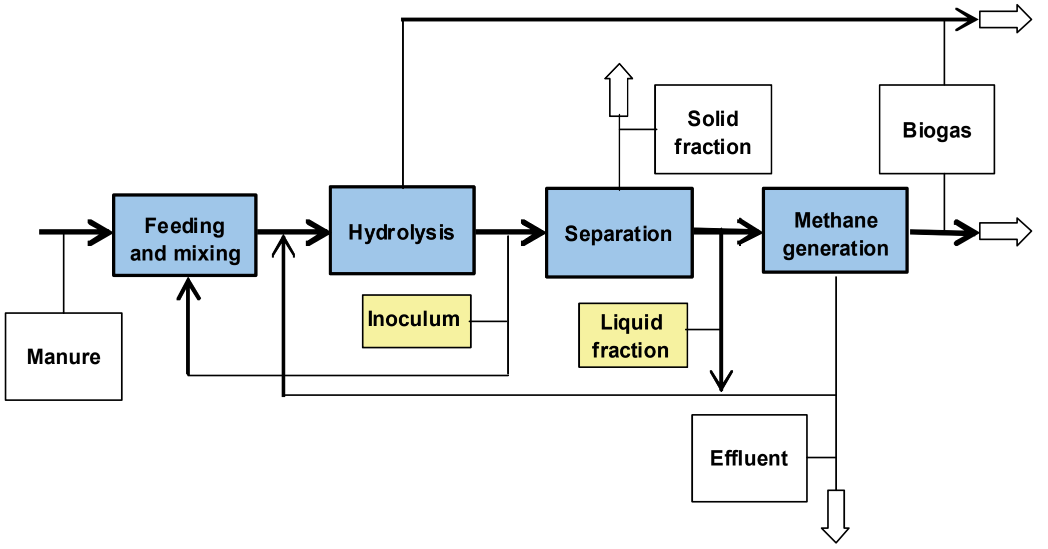

Fig. 5. Block diagram of the material flow of the prototype plant in Järna, Sweden

In a two-phase process, the hydrolysis reactor is continuously filled and discharged automatically. The output from the hydrolysis reactor is separated into a solid and liquid fraction. The solid fraction is composted. The liquid fraction is further digested in a methane reactor and the effluent is used as liquid fertiliser. The different process steps are described according to Figure 6.

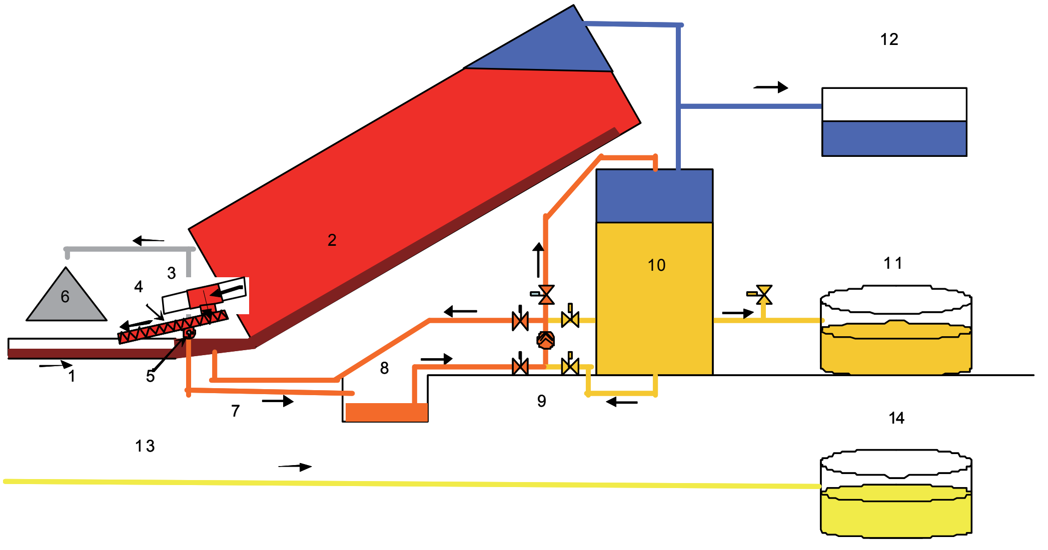

Fig. 6. Principle of operation of the prototype biogas plant at Yttereneby farm, Järna, Sweden. 1 feeder channel, 2 first or hydrolysis reactor, 3 drawer, 4 drawer discharge screw, 5 solid residue separation screw, 6 solid residue after hydrolysis, 7 drain pipe of liquid fraction, 8 liquid fraction buffer store, 9 pump and valve, 10 second or methane reactor, 11 effluent store, 12 gas store, 13 urine pipe, 14 urine store

A hydraulic powered scraper shifts manure of a dairy stanchion barn into the feeder channel (1) of the hydrolysis reactor (2). The manure is a mixture of faeces, straw and oat husks. The urine (13) is separated in the stall via a perforated scraper floor and stored separately (14). From the feeder channel the manure is pressed via a feeder pipe to the top of the 30° inclined hydrolysis reactor of 53 m^3 capacity. The manure mixes with the substrate sinking down by gravity force. After 22 to 25 days retention at 38 °C, a bottomless drawer (3) from the lower part of the reactor discharges the substrate. Every drawer cycle removes about 0.1 m^3 substrate from the hydrolysis reactor to be discharged into the transport screw (4) underneath. From the transport screw, the substrate partly drops into a down crossing extruder screw (5) where it is separated into solid (6) and liquid (7) fractions. The remaining material is conveyed back to the feeder channel and inoculated into the fresh manure.

The solid fraction from the extruder screw is stored at the dung yard for composting. The liquid fraction is collected into a buffer store (8) and from there pumped into the methane reactor (10) with 17 m^3 capacity. The methane reactor is 4 m high and filled with about 10,000 filter elements offering a large surface area for methane bacteria settlement. Liquid from the buffer and from the methane reactor partly returns into the feeder pipe to improve the flow ability. After 15 to 16 days retention at 38 °C the effluent is pumped into slurry store (11) covered by a floating canvas. A screw pump (9) conveys all liquids, directed by four pressurized air-driven valves. The gas generated in both reactors is collected and stored in a sack (12).

A compressor generates 170 mbar pressure to supply the burners of the process and estate boiler with biogas for heating purposes. A programmable logic controller regulates the biogas plant automatically.

4.3 Experimental results

The plant produced in average 52 m^3 biogas per day. Maximum yield was 91 m^3 biogas per day or 0.17 m^3 $CH4$/kg VS. From oat husks and straw, originate 53 to 70% of the organic dry matter of the input material. In the solid fraction remained 70 to 75% of the total solids, in the effluent 10 to 15% and within the biogas 14.8 to 14.9%. Because the solid fraction is removed after digestion of the manure in the first reactor, the loading rate and the yield rate cannot be calculated for the whole plant. This methodical problem makes it difficult to compare this plant with one-stage plants.

The volume efficiency of the plant is slightly better than the average of common slurry fermenters. An evaluation of on farm biogas plants (Bundesforschungsanstalt für Landwirtschaft (FAL), 2006) reported that 70% of the evaluated plants achieved a volume efficiency of 250 to 750 L biogas per m^3 and day. Up to 305 kWh per day or 56% of the produced energy was available for heating the farm estate. Composted solid fraction and effluent together contained 70 to 81% of the total input nitrogen and 94 to 111% of input NH_4.

The two-phase prototype biogas plant in Järna is suitable for digestion of organic residues of the farm and the surrounding food processing units. The plant works full-automatically. However, the two-phase process consumes much energy and the investment costs are high. There is still a lack of appropriate technical solutions in terms of handling organic material of high dry matter content, and process optimisation. The innovative continuously feeding and discharging technique is appropriate for the consistency and the dry matter content of the organic residues of the farm. It is probably not suitable for larger quantities of unchopped straw or green cut.

Reactor		R1	R2	R1 + R2	R1	R2	R1 + R2
Observation period		spring			autumn		
Effective capacity	m^3	53	18	71	53	18	71
Fresh mass input	kg/day	2,000	1,045	2,000	2,430	1,184	2,430
Specific weight input	kg/m^3	946	968		989	1,015	
Organic dry matter VS	kg/day	340	61	340	375	35	375
Organic dry matter	%	17	5.8	17	15	3	15
Retention time	days	25	16		22	15	
Loading rate	$kg/(m^3 day)$	6	3		7	2	
Biogas yield	L/kg VS	85	313	141	125	147	139
Methane yield	L/kg VS	48	204	85	71	96	80
Volume efficiency	$L/(m^3 day)$	544	1,093	681	887	297	740

Table 4. Performance parameters of the biogas prototype plant in Järna

Further pros and cons of the presented AD plant in Järna are compiled in Table 5 (it needs to be taken into account, that optimization was not yet fully completed).

Pros	
Operating	Full-automatically digestion of solid manure, no mixing required
Heat energy	Up to 1.7 kWh / kg organic dry matter, up to 57% energy surplus
N_{tot} losses	Up to 39% reduced compared to aerobic treatment
NH_4 losses	Up to 93% reduced compared to aerobic treatment
CH_4 generation	Up to 64% from oat husks (residues from the food processing unit) and straw
Cons	
Gas production	Average gas yield too far from the maximum yield
Heat consumption	Organic material must be heated twice
Investment costs	>2000 € per m^3 reactor volume

Table 5. Pros and cons of the prototype biogas plant in Järna, Sweden

Up to now, the technique of the prototype does not offer competitive advantages in biogas production compared to slurry based technology as far as only energy production is concerned. The results show that the ideal technical solution is not invented yet. This fact may be a challenge for farmers and entrepreneurs interested in planning and developing future competitive biogas plants on-farm suitable for solid organic matter.

5. Conclusions

Dry digestion of organic residues is particularly well suited and state-of-the art for treating the organic fraction of MSW. Segregation at source is among the main factors towards wider dissemination of this technology, and therefore regulatory frameworks are most important.

Dry digestion is less common in the agricultural sector, but the technology has experienced increasing interest in the last years, and it is to be expected that more dry digestion plants will be build.

Development of new prototype biogas plants requires appropriate compensation for environmental benefits like closed nutrient cycle and production of renewable energy to improve the economy of biogas production. The prototype in Järna described in Section 4 of this book chapter meets the set objectives since - beside renewable heat energy - a new compost product from the solid fraction is generated. However, the two-phase process consumes much energy and the investment costs are high.

Batch anaerobic dry digestion in box type fermenters promises further application in agriculture and for treatment of municipal solid waste, especially with smaller substrate throughputs. Methane yields can be achieved which are at the same level than the yields in wet digestion systems. A higher risk of inactive zones with inhibited biodegradation was, however, observed at full scale. This may be explained as result of lack of mixing during fermentation and due to inhomogeneous conditions over the substrate stack height.

For discontinuous digestion with sprinkling of process water, structure-rich biomass, e.g. green cut, landscape conservation residues or solid dung, is especially advantageous choice when considering process technology. In order to maximize gas production per reactor volume, mixtures of fractions with high energy content and structure-rich fractions are advisable.

6. Acknowledgements

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8

Production of Activated Char and Producer Gas Sewage Sludge

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1. Introduction

According to the depletion of fossil fuel and global warming, energy conversion technology for waste has been considered as value added alternative energy source. Among the potential waste that can be converted into energy, waste sludge continues to be increased due to increased amount of waste water treatment facilities, resulting from industry development and population increase. Most of waste sludge was treated through landfill, incineration, and land spreading (Fullana et al, 2003; Inguanzo et al, 2002; Karayildirim et al, 2006). However, landfill requires the complete isolation between filling site and surrounding area due to leaching of hazardous substance in sludge, and has the limited space for filling site. Utilization of sludge as compost incurs soil contamination by increasing the content of heavy metal in soil, and causes air pollution problem due to spreading of hazardous component to atmosphere. Incineration has the benefits of effective volume reduction of waste sludge and energy recovery, but insufficient mixing of air could discharge hazardous organic pollutant especially in the condition of low oxygen region. In addition, significant amount of ashes with hazardous component will be created after incineration.

As alternative technology for the previously described sludge treatment methods, researches on pyrolysis (Dominguez et al, 2006; Fullana et al, 2003; Karayildirim et al, 2006) and gasification treatment (Dogru et al, 2002; Phuphuakrat et al, 2010) have been conducted. Pyrolysis/gasification can produce gas, oil, and char that could be utilized as fuel, adsorber and feedstock for petrochemicals. In addition, heavy metal in sludge (excluding cadmium and mercury) can be safely enclosed. It is treated at the lower temperature than incineration so that amount of contaminant is lower in pyrolysis gasification gas due to no or less usage of air. Moreover, hazardous components, such as dioxin, are not generated. However utilization of producer gas from pyrolysis gasification into engine and gas turbine might cause the condensation of tar. In addition, aerosol and polymerization reaction could cause clogging of cooler, filter element, engine inlet, etc (Devi at el, 2005; Tippayawong & Inthasan, 2010).

As the reduction methods of tar component, in-pyrolysis gasifier technology (IPGT) and technology after pyrolysis gasifier (TAPG) were suggested. Firstly, IPGT does not require the additional post-treatment facility for tar removal, and further development is required for operating condition and design of pyrolysis gasifier. Through these conditions and technical advancement, production of syngas with low tar content can be achievable, but cost and large scaled complex equipments are needed (Bergman et al, 2002; Devi et al, 2003).

Secondly, multi-faceted researches on TAPG, such as thermal cracking (Phuohuakrat et al, 2010; Zhang et al, 2009), catalysis (Pfeifer & Hofbauer, 2008), adsorption (Phuohuakrat et al, 2010), steam reforming (Hosokai et al, 2005; Onozaki et al, 2006; Phuohuakrat et al, 2010), partial oxidation (Onozaki et al, 2006; Phuohuakrat et al, 2010), plasma discharge (Du et al, 2007; Guo et al, 2008; Nair et al, 2003; Nair et al, 2005; Tippayawong & Inthasan, 2010; Yu et al, 2010; Yu et al, 2010), etc have been conducted. For thermal cracking, higher than 800°C is required for the reaction, and its energy consumption surpass the production benefit. Catalyst sensitively reacts with contaminants such as sulfur, chlorine, nitrogen compounds from biomass gasification. Also, catalyst can be de-activated due to cokes formation, and additional energy cost to maintain high temperature is needed. For adsorption, there were several researches utilizing char, commercial activated carbon, wood chip and synthetic porous cordierite for tar adsorption. In case of adsorbers having mesopore, adsorption performance of light PAH tars, such as naphthalene, anthracene, pyrene, etc excluding light aromatic hydrocarbon tar (benzene, toluene, etc) was superior.

Tar reduction in steam reforming, partial oxidation and plasma discharge can produce syngas having major compounds of hydrogen and carbon monoxide through reforming and cracking reaction. The steam reforming has a good characteristic in high hydrogen yield. But it requires high temperature steam which consumes great deal of energy. In addition, longer holding time might require larger facility scale. On the contrary, partial oxidation reforming features less energy consumption, and has the benefit of heat recovery due to exothermic reaction. However, hydrogen yield is relatively small, and large amount of carbon dioxide discharge is the disadvantage. Researches on tar decomposition via plasma discharge were conducted in dielectric barrier discharge (DBD) (Guo et al, 2008), single phase DC gliding arc plasma (Du et al, 2007; Tippayawong & Inthasan, 2010; Yu et al, 2010), and pulsed plasma discharge (Nair et al, 2003). Compared to conventional thermal and catalytic cracking, the plasma discharge shows the higher removal efficiency due to the formation of radicals. However, high cost of preparation of power supply and short life cycle is the key for improvement. A 3-phase arc plasma applied for tar removal is easy to control the reaction, and has high decomposition efficiency along with high energy efficiency. That is to say; all the methods have limitation in the waste sludge treatment for producing products and removing tar in the producer gas. Therefore, the combination of both IPGT and TAPG should be accepted as a new alternative method for with feature of environment-friendliness.

In this study, thermal treatment system with pyrolysis gasifier, 3-phase gliding arc plasma reformer, and sludge char adsorber was developed for energy and resource utilization of waste sludge. A pyrolysis gasifier was combined as screw pyrolyzer and rotary carbonizer for sequential carbonization and steam activation, and it produced producer gas, sludge char, and tar. For the reduction of tar from the pyrolysis gasifier, a 3-phase gliding arc plasma reformer and a fixed adsorber bed with sludge char were implemented. System analysis in pyrolysis gasification characteristics and tar reduction from the thermal treatment system were achieved.

2. Experimental apparatus and methods

2.1 Sludge thermal treatment system

A pyrolysis gasification system developed in this study was composed of pyrolysis gasifier, 3-phase gliding arc plasma reformer, and fixed bed adsorber, as shown in figure 1.

A pyrolysis gasifier was designed to be a combined rig with a screw carbonizer for pyrolysis of dried sludge and a rotary activator for steam activation of carbonized material. The screw carbonizer was manufactured as feed screw type for carbonization of dried sludge. Feed screw controls the holding time of dried sludge at carbonizer according to motor revolution number. The screw carbonizer features dual pipe, and steam holes were installed at radial direction of external wall, and high pressure steam is discharged to activator radially. The rotary activator is composed of rotary drum with vane and pick-up flight, indirect heating jacket, pyrolysis gas outlet, gas sampling port, char outlet, etc. Retention time of activated sludge is controlled via number of rotation for a rotary drum. A sludge feeding device is for holding of dried sludge in a dried sludge hopper which is installed at inlet of the combined pyrolysis gasifier. A screw feeder is installed at the bottom of the hopper, and controls the input amount of dried sludge via revolution number. The feeder feeds the dried sludge into the screw carbonizer. A hot gas generator is for producing hot gas to heat a heating jacket and supplys hot steam into a rotary drum. It was composed of a combustor with burner and a steam generator.

A 3-phase gliding arc plasma reformer was installed at downstream of outlet for the pyrolysis gasifier. The gliding arc plasma reformer utilized a quartz tube (55 mm in diameter, 200 mm in height) for insulation and monitoring purposes, and a ceramic connector (Al_2O_3, wt 96%) in electrode fixing was adopted for complete insulation between three electrodes. The three conical electrodes in 120° (95 mm in length) were installed, maintaining 3 mm gap. At the inlet of the plasma reformer, a orifice disc with 3 mm hole for injection of producer gas was installed. A 3-phase AC high voltage power supply unit (Unicon Tech., UAP-15K1A, Korea) was used for stable plasma discharge at the inside of the plasma reformer.

A sludge char adsorber was made of a fixed bed cylinder (76 mm in diameter, 160 mm in length), and installed at the rear section of the plasma reformer. To fix the packing material at an adsorber, a porous distributer in stainless steel (25-mesh) was installed at the upper part. The porous distributer was made in a honeycomb ceramic for preventing channeling effect of input producer gas.

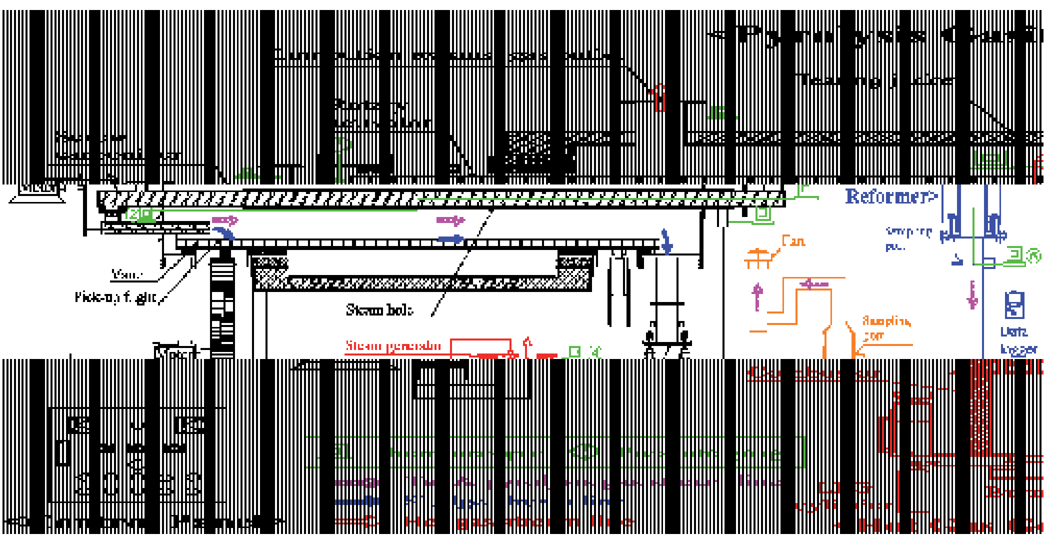

Fig. 1. Experimental setup of a pyrolysis gasification

Experiment was conducted at optimal condition for high quality porosity in sludge char and for the largest amount of combustible gas formation. The experimental conditions and each temperature condition were given in table 1. All the data in experiments were taken after stabilizing temperatures at each part, particularly the screw carbonizer and rotary activator. After finishing experiment by setting condition, sludge char in a char outlet is cooled up to room temperature by nitrogen passed the pyrolysis gasifier to protect the oxidation of the sludge char by air. Gas was sampled for 5 minutes in a stainless cylinder at the sampling ports of each pyrolysis gasifier, plasma reformer, and adsorber (Refer a gas sampling line in section 2.3.2). For tar sampling, it was conducted for 20 minutes by tar sampling method (as shown section 2.2), and total amount of gas was measured with a gas-flow meter. For a test, the gas and tar sampling were conducted 3 times during test time of 120 minutes stably, and the taken data were averaged. Adsorption capacity of sludge char was calculated from weight of adsorber before/after experiment divided by test time.

<table>
<tr><td colspan="6">Test conditions</td></tr>
<tr><td colspan="2" rowspan="2">Steam feed amount (mL/min)</td><td colspan="2" rowspan="2">Moisture content of dried sludge (%)[1)]</td><td colspan="2">Retention time (min)</td></tr>
<tr><td>Activator</td><td>Carbonizer</td></tr>
<tr><td colspan="2">10</td><td colspan="2">9.8</td><td>30</td><td>30</td></tr>
<tr><td colspan="6">Temperature (℃) in each part</td></tr>
<tr><td>①Combustor</td><td>②Carbonizer</td><td>③Activator</td><td>④Steam generator</td><td>⑤Plasma reformer</td><td>⑥Adsorber</td></tr>
<tr><td>1,010</td><td>450</td><td>820</td><td>450</td><td>400</td><td>35</td></tr>
</table>

1) Moisture content of dried sludge is average number

Table 1. Detailed conditions in each section

2.2 Tar sampling and analysis methods

Tar sampling and analysis were used by the method of biomass technology groups (BTGs) (Good et al, 2005; Neeft, 2005; Phuohuakrat et al, 2010; Son et al, 2009; Yamazaki et al, 2005). Wet sampling module was installed with 6 impingers (250 mL) in two separated isothermal baths for adsorption of tar and particles. At the first isothermal bath, 100 mL of isopropanol was filled into 4 impingers, respectively, along with 20°C of water. For the second bath, isopropanol was filled while it was maintained at -20°C using mechanical cooling device (ECS-30SS, Eyela Co., Japan). Among 2 impingers, 1 unit was filled with 100 mL of isopropanol, and the other was left as empty. In the series of impinger bottles, the first impinger bottle acts as a moisture and particle collector, in which water, tar and soot are condensed from the process gas by absorption in isopropanol. Other impinger bottles collect tars, and the empty bottle collects drop.

Immediately after completing the sampling, the content of the impinger bottles were filtered through a filter paper (Model F-5B, Advantec Co., Japan). The filtered isopropanol solution was divided into two parts; the first was used to determine the gravimetric tar mass by means of solvent distillation and evaporation by evaporator (Model N-1000-SW, Eyela, Japan) in which temperature and steam pressure were 55~57°C and 230 hPa, respectively. The second was used to determine the concentrations of light tar compounds using GC-FID (Model 14B, Shimadzu, Japan).

Quantitative tar analysis was performed on a GC system, using a RTX-5 (RESTEK) capillary column (30 m - 0.53 mm id, 0.5 μm film thickness) and an isothermal temperature profile at

45°C for the first 2 min, followed by a 7 °C/min temperature gradient to 320°C and finally an isothermal period at 320°C for 10 min. Helium was used as a carrier gas. The temperature of the detector and injector were maintained at 340 and 250°C, respectively.

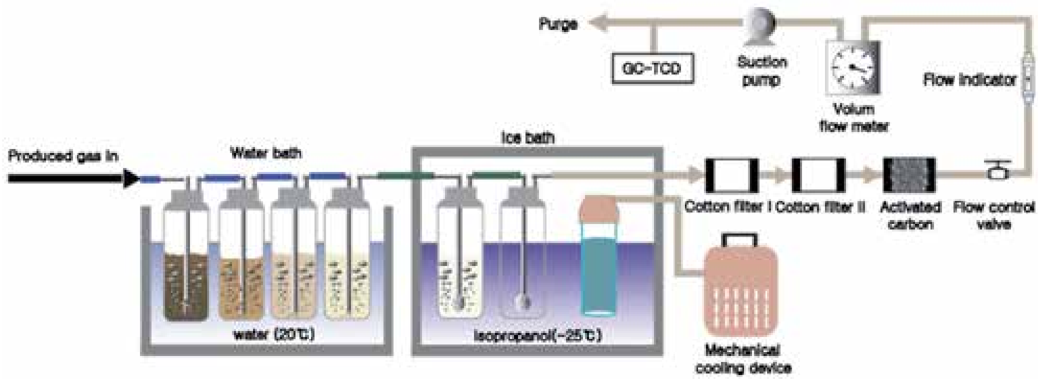

Fig. 2. Tar sampling line system

2.3 Sludge char and gas analysis

2.3.1 Pore development in sludge char

The structural characterization of the sewage sludge char was carried out by physical adsorption of N_2 at -196°C. The adsorption isotherms were determined using nanoPOROSITY (Model nanoPOROSITY-XQ, MiraeSI Co. Ltd, Korea). The surface area was calculated using the BET (Brauner-Emmet-Teller) equation. Using BJH (Barret-Joyner-Halenda) equation, incremental pore volume and mean pore size was calculated. To compare pore development in sludge char, SEM (scanning electron microscopy; Model S-4800, Hitachi Co., Japan) was used, and image was taken at 50,000X resolution for morphological analysis. Chemical properties and constituent components were analyzed via EDX (Energy-dispersive X-ray spectroscopy; Model 7593-H, Horiba, UK).

2.3.2 Sampling and analysis producer gas

The produced gas was sampled for 5 minutes in a stainless cylinder as sampling gas flow rate is 1 L/min. As can be seen in figure 2, a set of backup VOC adsorber was installed downstream of the series of impinger bottles to protect the column of the gas chromatography from the residual solvent or VOCs, which may have passed through the impinger train. The set of backup VOC adsorber consists of two cotton filters and an activated carbon filter connected in a series. Gas analysis was conducted with GC-TCD (Model CP-4900 Varian, Netherland). MolSieve 5A PLOT column for H_2, CO, O_2, and N_2 and PoraPLOT Q column for CO_2, CH_4, C_2H_4, and C_2H_6 were used for simultaneous analysis.

3. Results and discussion

3.1 Dried sludge characteristics

Sludge from a local wastewater treatment plant was dewatered by a centrifuging. And then the dewatered sludge was dried to less than 10% of moisture content using a rotary kiln type dryer developed by the corresponding researcher. The pyrolysis gasification is a

process of which heat is applied by external source or partial oxidation. Vaporization temperature of moisture is lower than thermal decomposition temperature for organic compound in sludge. Therefore, high moisture content in sewage sludge will show significant energy loss due to preemptive utilization of the heat for drying.
In addition, delayed pyrolysis gasification will affect the producer via reaction with moisture and reactant. Therefore, less than 10% of moisture content in the dried sludge was taken for this study.
Table 2 shows proximate analysis and ultimate analysis on the dried sludge.

Proximate analysis (%)				
Moisture	Volatile matter	Fixed carbon	Ash	
9.7	51.7	6.1	32.5	
Ultimate analysis (%)				
C	H	O	N	S
52.3	8.2	32.2	7.92	0.01

Table 2. Properties of the dried sludge

3.2 Thermal behavior analysis

To determine pyrolysis temperature, TGA (thermo gravimetric analysis) and DTG (derived thermo-gravimetric) analysis was shown in figure 3. According to TGA and DTG results, the maximum weight loss temperature and final decomposition temperature, etc can be derived (Karayildirim et al, 2006).

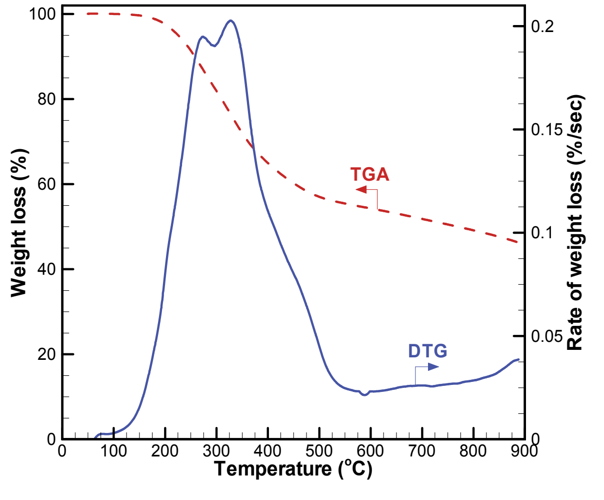

Fig. 3. TGA and DTG for pyrolysis of the dried sewage sludge

Thermal decomposition of the dried sludge showed weight loss after evaporation of small moisture content at 100~150°C as shown in DTG curve. This could be elucidated by two steps. First step (primary pyrolysis) is discharging of volatile component at 200~500°C, and the second step is decomposition of inorganic compound at over 500°C. First step for volatile component discharge displayed two peaks, and it can be explained as follows. The first peak might be due to decomposition and devolatilization of less complex organic structures which is a small fraction. The second peak was caused by decomposition of more complex organic structures corresponding to a larger fraction. Second step (secondary pyrolysis) is related to decomposition of inorganic compound as described before. In first step, TGA displayed 57% at 500°C, and 900°C for the second step was 46.2%. That is, 43% of moisture content and volatile component was discharged during the first step, and in second step 10.8% reduction (from first step) was corresponded to decomposing ash which is an inorganic component in dried sludge.Therefore, for the pyrolysis gasification experiment in purpose of improved yield of producer gas and higher adsorption rate, pyrolysis carbonization were maintained at 450°C which discharges the largest amount of volatile component, and steam activation was set to 850°C for increasing the porosity in the sludge char.

3.3 Characteristics of a pyrolysis gasifier

Figure 4 shows mass yield for char, tar, and gas from a pyrolysis gasifier. The product amount ordered was producer gas of 43.6%, sludge char of 35.4% and tar of 21%. As described before, the corresponding experiment setup was made to primary pyrolysis carbonization at screw carbonizer which is set to 450°C and post-activation at rotary kiln activator along with steam injection, which is set to 820°C.

Producer gas was formed by decomposition and volatilization of organic compound in a screw carbonizer (refer first step description of DTG in figure 3), and gas formation was increased due to steam reforming of tar and char in a rotary kiln activator. Sludge char in mass was reduced by vaporization of volatile component during the passing of the carbonizer, and steam gasification and inorganic decomposition in the activator. Heavy tar was formed and then it was converted into producer gas and light tar at the activator.

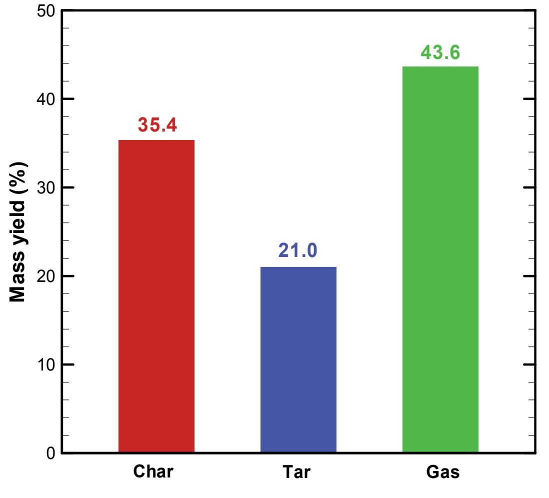

Fig. 4. Mass yield of the products

3.3.1 Characteristics of the sludge char

Figure 5 compares incremental pore volume and SEM photos of the dried sludge and sludge char. The pore size classification in this study follows the IUPAC classification (IUPAC, 1982; Lu, 1995) i.e. micropores (<20 Å), mesopores (20~500 Å) and macropores (>500 Å). Pore

of sludge char after carbonization activation showed significant increase compared to the dried sludge, and pore distribution was less than 500 Å, which is comprised of micropores and mesopores. The pyrolysis gasifier in this study had been designed as continuously combined type for carbonization of dried sludge at a screw carbonizer and steam activation at a rotary activator. The dried sludge experienced evaporating of moisture and decomposing of organic component for pore development through passing the screw carbonizer (Lu, 1995). And then carbonized material was exposed to steam at the rotary activator for the formation and development of micropoees and mesopores. For steam activation in developing micropores, steam should deeply penetrate into pores of the carbonized material for surface reaction. High temperature activation had the benefit of diffusion and penetration of the steam to develop micropore. On the other hand, it was blocked by tar in the carbonized material, resulted as well-developed mesopore. This is the reason that the sludge char from the carbonization activation had well-developed micrepores and mesopores. Sludge drying was made with a parallel flow rotary kiln drier with direct-hot gas application. Hot gas inflow in turbulent flow was directly contacted with the dewatered sludge in the dryer. Inside of the dryer was set to 255°C in average value.
For dried sludge, small portion of micropore and mesopore was formed. It is considered to be formed due to discharging of volatile organic material and dehydroxlation of inorganic material from the dried sludge. Bagreev et al. proved that water released by the dehydroxylation of inorganic material could aid pore formation and moreover could act as an agent for creating micropores (Bagreev et al, 2001). In addition, Inguanzo et al. proposed that carbonization increases the porosity through unblocking many of the pores obscured by volatile matter (Inguanzo et al, 2001). Surface of the dried sludge from SEM photograph in 50,000 times of magnification shows smooth surface with less pores, but the sludge char presents overall formation of pores.

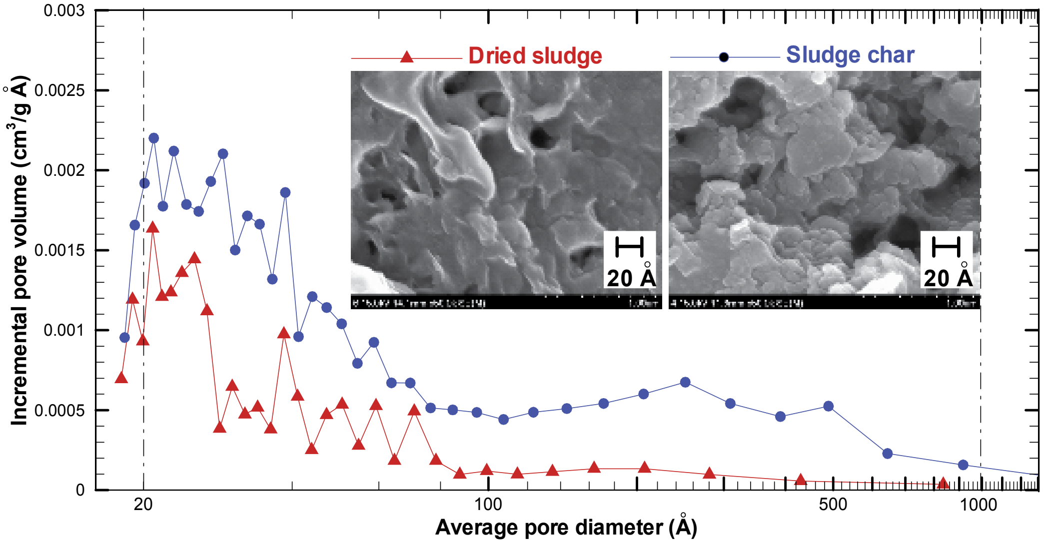

Fig. 5. Incremental pore volume and SEM images of the dried sludge and sludge char

Table 3 compares the results of the sludge char made from this study and 3 types adsorbent from the study of Thana Phuphuakrat etc (Phuphuakrat et al, 2010). For the sludge char,

specific surface area and pore volume were smaller than commercial activated carbon, and mean pore size was larger. The sludge char displayed mesopore similar to wood chip and synthetic porous cordierite, but the activated carbon featured micropore.
Adsorption capability of the sludge char was less than the one with wood chip, but larger than the one of activated carbon and synthetic porous cordierite. The adsorption experiment in this study was conducted by using benzene only. So the comparison in the adsorption capacity has difficulty because the study of Thana Phuphuakrat etc was achieved in a continuous test rig using Japanese cedar which produced pyrolysis gas including all tars and water. However, it might be considered that the wood chip adsorbed large amount of steam when compared to the sludge char, because of hydrophilic surface and mesoporous material favoring water adsorption. Although the test in non-condensable light aromatic hydrocarbon (e.g. benzene) was conducted in this study, it should be expected for the sludge char to adsorb well for the condensable light PAH (e.g. naphthalene, anthracene, pyrene) due to having mesopores as proved in the other study.

Adsorbent	Specific surface area (m^2/g)	Mean pore size (Å)	Pore volume (cm^3/g)	Adsorption capacity (mg/g)
Sludge char[1)]	98.1	63.49	0.2354	120.6
Activated carbon	987.1	11.28	0.5569	97.5
Wood chip	1.072	100.77	0.0058	155.7
Synthetic porous cordierite	6.045	27.43	0.0083	12.8

1) Sludge char from this study

Table 3. Porous characteristics and adsorption capacity of the adsorbents from this study and other results (Phuohuakrat et al, 2010)

A semi quantitative chemical analysis of dried sludge and sludge char, figure 6 and table 4, was obtained from the EDX analyzer coupled to SEM measurements. The results indicate that both samples present relatively high carbon content in addition to mineral components. The relative amount of carbon decreased after carbonization and activation, as expected considering the decomposition of the organic components.
These atoms might be considered as the potential catalysts for pyrolysis reaction. For example, with Al, if existing in the form of Al_2O_3, it would be an acid catalyst for cracking reaction (Sinfelt & Rohrer, 1962); or with K, and Ca atoms, they were already reported as the catalyst for biomass pyrolysis in literature (Yaman, 2004).
Figure 7 shows the N_2 adsorption-desorption isotherm for the dried sludge and sludge char. According to the isothermal adsorption graphs, the dried sludge exhibited only a small amount of adsorption, but the sludge char displayed a larger amount of adsorption at lower nitrogen concentrations. As shown in figure 5, the sludge char exhibited well-developed micro- and meso-pore structures. The analysis on the adsorption isotherm provides an assessment for the pore size distribution. According to the IUPAC classification, the curve of the sludge char corresponds to Type V isotherm. A characteristic of the Type V isotherm is the hysteresis loop, which is associated with the capillary condensation in mesopores and limiting uptake at high relative pressure (Khalili et al, 2000).

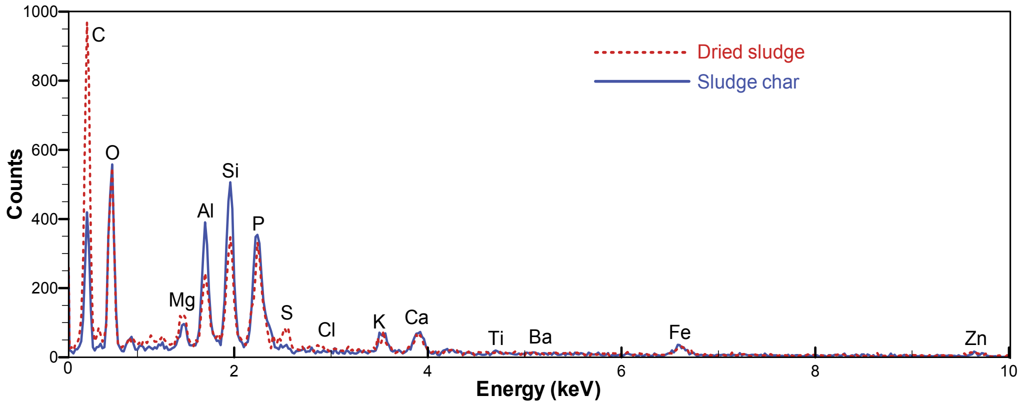

Fig. 6. EDX spectrums of dried sludge and sludge char

Item	C	O	Mg	Al	Si	P	S	Cl	K	Ca	Ti	Fe	Zn	Ba
Dried sludge (wt %)	53.65	44.62	0.06	0.23	0.45	0.55	0.03	0.01	0.06	0.07	0.01	0.24	0.02	0
Sludge char (wt %)	47.65	44.83	0.14	1.21	5.34	0.46	0.03	0.02	0.09	0.11	0	0.21	0	0.01

Table 4. Elements content of dried sludge and sludge char

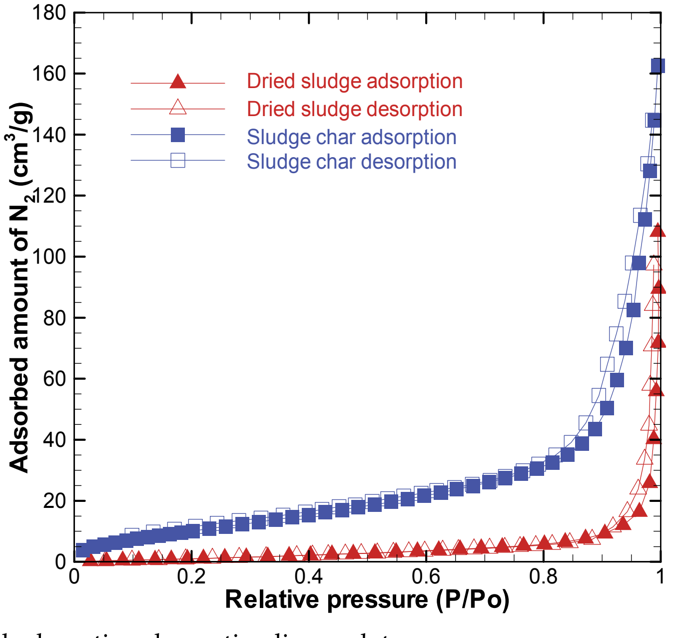

Fig. 7. Isothermal adsorption-desorption linear plot

3.3.2 Tar characteristics for the pyrolysis gasification

Results produced from the pyrolysis gasifier were shown in table 5.

Representative tars for the corresponding benzene ring were selected to benzene (1 ring), naphthalene (2 ring), anthracene (3 ring) and pyrene (4 ring). And the representative tars with nitrogen for the sewage sludge (Fullana et al, 2003) were taken as benzonitrile and benzeneacetonitrile. Gravimetric tar was 26.3 g/Nm^3. Total concentration of light tar was 10.9 g/Nm^3, and its amount order was benzene, naphthalene, benzonitrile, benzeneacetonitrile, anthracene, and pyrene. Dried sludge formed sludge char, tar, and gas during pyrolysis at screw carbonizer, and then steam activation was achieved in rotary activator. The gravimetric tar is total amount of tar after passing carbonization and activation process. Benzene and naphthalene among light tar are products produced during secondary pyrolysis at carbonizer, and some part of both tars converts to gas during steam activation at activator. In addition, anthracene and pyrene were directly formed by primary pyrolysis from dried sludge at carbonizer. Both tars should be known as not affecting by carbonization-activation temperature and steam amount (Umeki, 2009).

Gravimetric tar	Benzene	Naphthalene	Anthracene	Pyrene	Benzo-nitrile	Benzene-acetonitrile
26.3	6.31	2.97	0.87	0.12	0.61	0.11

Table 5. Tar concentrations from a pyrolysis gasifier (unit: g/Nm^3)

3.3.3 Producer gas characteristics

Table 6 shows producer gas concentration and higher heating value from a pyrolysis gasifier. Major components in gas were analyzed to be hydrogen, carbon monoxide, methane, and carbon dioxide along with trace amount of nitrogen and oxygen. The higher heating value was 13,400 kJ/Nm^3 having half value of natural gas.

H_2	CO	CH_4	CO_2	C_2H_4	C_2H_6	O_2	N_2	Higher heating value
41.2	17.3	9.5	15.4	0	0	0.5	3.3	13,400

Table 6. Concentration of producer gas (dry vol. %) and higher heating value (kJ/Nm^3)

Hydrogen was produced by the cracking of the volatile matter generated by the pyrolysis gasification. Methane resulted from cracking and depolymerization reactions, while carbon monoxide and carbon dioxide were produced from decarboxylation and depolymerization or the secondary oxidation of carbon (Xiao et al, 2010). In addition, the presence of steam at high temperatures gave rise to in situ steam reforming of the volatile matters and partial gasification of the solid carbonaceous residue, as shown in the reactions of Eqs. (1) and (2). Non-condensable products may also undergo gas phase reactions with each other. For example, the CO and CH_4 contents may be affected by the methane gasification and water gas shift reactions, as shown in Eqs. (3) and (4) (Domínguez et al, 2006).

- Steam reforming reaction:

$$Organics(g) + H_2O(g) \leftrightarrow CO + H_2 \tag{1}$$

- Steam gasification reaction:

$$C(s) + H_2O(g) \leftrightarrow CO + H_2,\ \Delta H_{298K} = 132 kJ\, mol^{-1} \tag{2}$$

- CH_4 gasification reaction:

$$CH_4 + H_2O \leftrightarrow CO + 3H_2, \quad \Delta H_{298K} = 206.1 kJ\, mol^{-1} \tag{3}$$

- CO shift reaction:

$$CO + H_2O \leftrightarrow CO_2 + H_2, \quad \Delta H_{298K} = -41.5 kJ\, mol^{-1} \tag{4}$$

High temperatures were also responsible for the reduction of C_2H_4, C_2H_6 and C_3H_8. Some of the typical reactions are as follows (Zhang et al, 2010):

$$C_2H_6 \rightarrow C_2H_4 + H_2 \tag{5}$$

$$C_2H_4 \rightarrow CH_4 + C \tag{6}$$

However, it should be noted that the gas composition may not exclusively be the result of tar cracking and the partial gasification of char due to the complicated interactions of the intermediate products, which would probably affect the final gas composition.

3.4 Plasma reformer and adsorber characteristics

The plasma reformer was installed for converting produced tar from the pyrolysis gasifier into syngas via decomposition and steam reforming. In addition, the fixed bed adsorber was implemented for adsorption of by-passed tar from the plasma reformer.

3.4.1 Tar destruction performance

Fig. 8 shows the results of tar sampling at the rear section of the pyrolysis gasifier, plasma reformer, and fixed bed adsorber. Gravimetric tar concentration at the outlet of carbonization activator was 26.3 g/Nm^3, and it was reduced to 4.4 g/Nm^3 at the reformer outlet. Decomposition efficiency of the corresponding gravimetric tar was 83.2%. For light tar, total amount of carbonization activator outlet was 10.9 g/Nm^3. The concentration was reduced to 1.3 g/Nm^3 at the outlet of reformer, and the destruction efficiency of the light tar was 87.9%. Each concentration of the light tars was found to be 0.62 g/Nm^3 for benzene, 0.45 g/Nm^3 for naphthalene, 0.14 g/Nm^3 for anthracene, 0.021 g/Nm^3 for pyrene, 0.08 g/Nm^3 for benzonitrile, and 0.015 g/Nm^3 for benzeneacetonitrile.

Decomposition of heavy tar was happened due to plasma cracking and carbon formation in Eqs. (7) and (8) (Tippayawong & Inthasan, 2010). In addition, steam in producer gas from the pyrolysis gasifier formed excitation species as shown in Eq. (9), and the radicals reduced light tar and carbon black which produce by the reactions of plasma cracking and carbon formation (Guo et al, 2008). It is remarkable that the tars from the pyrolysis gasification should be decomposed significantly by the plasma reformer.

- Plasma cracking:

$$pC_nH_x \rightarrow qC_mH_y + rH_2 \tag{7}$$

- Carbon formation:

$$C_nH_x \rightarrow nC + (x/2)H_2 \tag{8}$$

- Water excitation:

$$H_2O \rightarrow H,\ e^-_{eq}, OH, H_2, H_2O_2, H_3O^+, Oh \tag{9}$$

In Eq. (9), C_nH_x represents tar, such as the large molecular compounds, and C_mH_y represents a hydrocarbon with a smaller carbon number compared to that of C_nH_x. Discharged residual tar from the plasma reformer was removed by the fixed bed adsorber filled with sludge char. Gravimetric tar at the adsorber outlet displayed 0.5 g/Nm³, which is 88.6% of removal efficiency. Total amount of light tar was 0.39 g/Nm³, which is corresponded to 40.5% of removal efficiency. The relevant concentration was 0.28 g/Nm³ for benzene, 0.09 g/Nm³ for naphthalene, 0.14 g/Nm³ for anthracene, 0.01 g/Nm³ for benzonitrile, and 0.003 g/Nm³ for benzeneacetonitrile. Among residual tar, heavy tar was mostly removed at adsorber, and non-condensed light tar that was not adsorbed was considered to be passed through the adsorber. For satisfactory IC engine operation, an acceptable particle content <50 mg/Nm³ and a tar content <100 mg/Nm³ is postulated (Milne et al, 1998). Therefore, 0.5 g/Nm³ of tar concentration in producer gas is sufficient for utilization. In addition, sampling analysis on particulate matter was not conducted in this study, but the carbon black was not formed due to steam reforming at the plasma reformer. Therefore, it is not considered to be problematic in the operation.

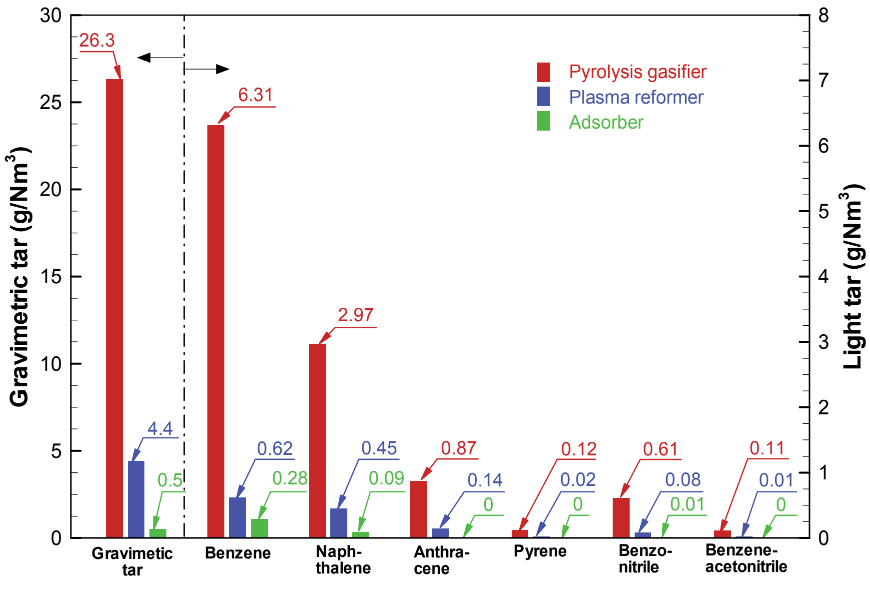

Fig. 8. Gravimetric tar and light tar concentrations

3.4.2 Gas formation characteristics

Figure 9 shows the producer gas analysis sampled from the pyrolysis gasifier, plasma reformer, and fixed bed adsorber, respectively. At the outlet of plasma reformer, gas concentration was found to be 50.9% for H_2, 22.3% for CO, 11% for CH_4, 8.7% for CO_2, 0.4% for C_2H_2, and 0.2% for C_2H_4. Concentration of hydrogen, carbon monoxide, and light hydrocarbon (methane, ethylene, and ethane) were increased compared to the outlet concentration of pyrolysis gasifier. For hydrogen and carbon monoxide, it was increased due to Eqs. (1) and (3), steam reforming and methane gasification reaction, respectively. Light hydrocarbon was converted from light tar using tar plasma cracking reaction (7) in portion and from chain reactions of Eqs. (5) and (6). In addition, decrease in carbon dioxide was considered to be dry reforming as shown in Eq. (10) (Devi et al, 2005).

$$C_nH_x + nCO_2 \rightarrow (x/2)H_2 + 2nCO \tag{10}$$

According to gas analysis results at adsorber outlet, 50.5% of H_2, 21.9% of CO, 10.5% of CH_4, 7.7% of CO_2, and 0.1% of C_2H_2 were displayed. Compared to the results at plasma reformer outlet, the corresponding concentration was slightly decreased within measurement tolerance, but it was not almost adsorbed. Higher heating value calculated using the gases from each outlet. It was found to be 11,200 kJ/Nm^3 for producer gas from the pyrolysis gasifier, 13,992 kJ/Nm^3 for the plasma reformer and 13,482 kJ/Nm^3 for the adsorber. The increase at the plasma reformer outlet is due to increased amount of combustible gases, particularly methane having high calorific value.

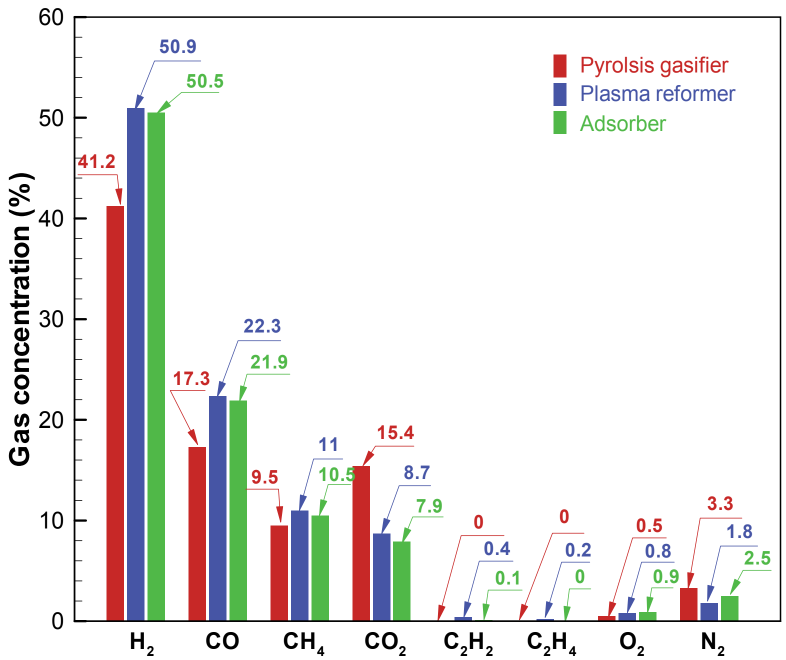

Fig. 9. Producer gas concentrations at exit of each part

4. Conclusions

To utilize dried sewage sludge as energy and resource, pyrolysis gasifier, plasma reformer, and fixed bed adsorber system were established. From the pyrolysis gasifier, sludge char and pyrolysis gases were produced along with small amount of tar. To improve tar adsorption capability of sludge char, an integrated pyrolysis gasifier was developed for achieving in sequential carbonization and activation. In addition, for higher producer gas yield and tar reduction, a plasma reformer was installed at the rear section of the pyrolysis gasifier, and a fixed bed adsorber, which contains sludge char from the pyrolysis gasifier, was implemented for adsorption of residual tars. Sludge char from the pyrolysis gasifier displayed 98.1 m^2/g of specific surface area and 63.49 Å of mean pore size, showing the distribution of mesopore and micropore with superior adsorption capability. Producer gas was mostly comprised of hydrogen, carbon monoxide, methane, and carbon dioxide, and the corresponding higher heating value was 13,400 kJ/Nm^3. Gravimetric tar was 26.3 g/Nm^3, and total amount of light tar was 10.9 g/Nm^3, which showed benzene, naphthalene, benzonitrile, and benzeneacetonitrile according to the concentration level. Plasma reformer featured tar cracking and steam reformation, and decomposition efficiency of gravimetric tar was 83.2%, which is corresponded to 4.4 g/Nm^3. For light tar, total amount was 1.3 g/Nm^3, which is 87.9% of decomposition efficiency. Hydrogen, carbon monoxide, and methane among the components of reforming gas were increased, having 13,992 kJ/Nm^3 of higher heating value. Gravimetric tar at the adsorber outlet was 0.5 g/Nm^3, which is 88.6% of decomposition efficiency. Total amount of light tar was 0.39 g/Nm^3, and it was 40.5% of decomposition efficiency. According to gas analysis results, 50.5% of H_2, 21.9% of CO, 10.5% of CH_4, 7.7% of CO_2, and 0.1% of C_2H_2 were displayed, and the corresponding higher heating value was 13,482 kJ/Nm^3. Therefore, carbonization-activation of sludge can form sludge char that could be utilized for tar adsorption, and the relevant clean producer gas is proved to be applicable for heat engine.

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9

Modelled on Nature – Biological Processes in Waste Management

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1. Introduction

Biological degradation and transformation of organic substances under aerobic or anaerobic conditions are key processes within the natural metabolism of an equilibrated circulation system in order to handle the accumulating biomass. These fundamental processes are the basis for management strategies focusing on the biological treatment of organic waste materials. They are subjected to the biochemical metabolism using the capability of microbial populations to degrade, transform and stabilise organic matter. Stabilisation comprises biological as well as abiotic chemical and physical processes and their interaction. Avoiding greenhouse gases and shortening the after care period stabilisation is the key target for safe waste disposal in landfills. Biogenic waste materials are a source of secondary products: biogas obtained by anaerobic digestion and composts produced under aerobic conditions. For composts stabilisation is a relevant process to achieve plant compatibility and persistent organic substances for soil amelioration. Biological processes additionally contribute to landfill remediation, e.g. by methane oxidation.

Nevertheless, biological degradation of waste materials is ambivalent and can lead to harmful effects if microbial activities take place under uncontrolled conditions in imbalanced systems. Abandoned landfills from the past demonstrate this fact. Anthropogenic organic wastes differ from "natural" organic waste by their amount, their heterogeneity and the content of xenobiotics. Therefore it is necessary to support and optimise biological degradation of waste organic matter by adequate process operation and technical devices. The equilibrium of necessary mineralisation and accessible humification is a topic of high interest in the context of carbon fixation.

"Optimisation" is no aspect in the context with natural degradation processes. Additionally they are not harmless a priori. They take place under the current conditions, but it can be assumed that an equilibrium is reached over longer periods of time. Changes of environmental conditions by anthropogenic activities can accelerate biological degradation. Peat bogs that were drained and amended with carbonates lose organic matter due to mineralisation (Küster, 1990). The pH value, water and air supply and temperature mainly influence the transformation rate. This fact indicates that biodegradability is not only an inherent property that depends on chemical and physical features of the material. The behaviour of biodegradable substances is affected by the interaction of both material characteristics and environmental conditions.

This chapter provides an overview of biological processes in waste management, targets and benefits, weak points and optimisation potential, process and product control by modern analytical tools such as FT-IR spectroscopy and thermal analysis.

2. Composting and anaerobic digestion - Environmental benefits of resource recovery

The biological treatment of waste materials primarily focuses on stabilisation of organic matter in order to avoid gaseous emissions after waste disposal. The aspect of resource recovery has gained in importance during the last two decades. Although resource recovery has been practiced in the past, e.g. by composting of organic residues, this idea is currently going through a renaissance, primarily due to the necessity of energy supply and increase of soil organic matter by compost application. The retrieval of chemical products from waste materials is also under discussion.

The knowledge about the biodegradability and microbial processes is a prerequisite for the optimum use of biogenic waste. The heterogeneous composition of the incoming material additionally demands a certain flexibility and adaptation according to basic requirements. In many cases there is a potential for process optimisation.

Soil improvement by compost application and its relevance to carbon storage and climate change

The benefits of compost application have been known for long time. According to historical traditions clever farmers recognised the value of "rotted" and "putrefying" organic waste for soil amelioration (Bruchhausen 1790, cited by Eckelmann, 1980). Compost management for many centuries has led to the formation of anthropogenic soils in several north-western European countries and in Russia (Hubbe et al., 2007). These so called "Plaggensoils" represent an impressive example of organic matter increase by compost application. "Terra preta" in the Amazon region also attests to the long-term effect of organic matter brought into soil by anthropogenic activities and organic waste (Sohi et al., 2009). Long-term experiments that have been initiated in the 19^{th} century provide useful data on the effects of organic matter amendments and their long-term behaviour (Jenkinson & Rayner, 1977).

Agricultural activities, tillage and the application of mineral fertilisers have promoted losses of organic matter in soils that have caused their degradation to a certain degree. "Desertification" has become a keyword in this context (Montanarella, 2003). The current issue of climate change has additionally attracted notice to carbon losses. The maintenance of organic matter and organic carbon is an effective measure to reduce CO_2 emissions. Besides technical approaches of carbon sequestration, prevention of carbon losses in soils by adequate tillage and compost application, which seems an effective measure should be given priority. Composts with high humic substance contents play a crucial role as they favour the fixation of carbon and minimise the losses.

How compost organic matter is integrated in different soil carbon pools is a topic of high interest in order to evaluate the stability and the long-term behaviour. Different approaches have been applied to identify and describe the carbon pools in soils (Six et al., 2000a; Six et al., 2000b; Pulleman & Marinissen, 2004). These methods can be applied to amended soils in order to trace the fate of compost organic matter and to quantify the contribution of composts to the stable carbon pool.

2.1 Composting

Composting is a biotechnological process that can be operated at different technical levels. Due to this fact composting is an appropriate technique for developing countries to handle biogenic resources for soil amelioration. Besides the environmental aspect resource recovery is a crucial issue. The application of composts on agricultural soils has gained in importance in view of the considerable losses of organic matter and soil degradation in many countries.

2.1.1 Regulations for compost quality - European and American situation

No European directive or regulation on compost quality determination has been put into force to date. A first step to establish such regulations was done by the Commission of the European Community in December 2008 by a green paper called "On the management of bio-waste in the European Union" (COM(2008) 811 final) (Commission of the European Communities, 2008). In this green paper national compost standards and legislations of the Member States are summarised. Compost policies and regulations differ substantially between the Member States. In Bulgaria, Cyprus, the Czech Republic, Denmark, Estonia, Hungary, Malta, Poland, Romania, Sweden and the United Kingdom no specific compost legislation exists. In Lithuania, France and Slovakia compost regulations were integrated in the waste and environmental legislation or only simple registration schemes were established. In Belgium, Finland, Germany and Austria specific compost standards are available. Austria, Belgium and Finland have an obligatory and Germany a voluntary quality assurance system. But only in Austria compost reaches the level of a product.

In Austria the "Compost Ordinance" (BMLFUW, 2001) was put into force in 2001. These rules defined limit values for pollutants (especially for heavy metals), foreign matter (plastics, glass, metals) and plant compatibility (maturity, toxic components). The Austrian Compost Ordinance provides three compost classes that are distinguished by both the input materials (e.g. kitchen, yard and market waste, sewage sludge) and the specific limit values for heavy metals. The compliance with the Austrian Compost Ordinance is supported by the „Ordinance for the separate collection of biogenic waste from households" (BMLFUW, 1992) which was enacted in 1992. It includes the obligation for the separate collection of biogenic waste from households, the recycling and use of these materials.

In America no directive or regulation on compost quality determination has been established up to date. The 50 federal states of America can rule compost quality by themselves. If there is any regulation available it only sets limit values for pollutants, especially for heavy metals.

2.1.2 Adequate ingredients and process operation

A wide range of organic waste materials is available. There are several synonymic terms to describe the waste fraction that serves as input material for anaerobic digestion and composting: organic waste, biogenic waste and biowaste are the most common ones. Besides yard and kitchen waste that have always been a basic component of composts, residues from food industry (Grigatti et al.; Bustamante et al., 2011) and biotechnological processes, agriculture, sewage sludge (Doublet et al., 2010), digestates from anaerobic processes and mixtures of these materials extend the list of ingredients for composting. Agricultural waste comprises crop residues and manure (Shen et al., 2011). Due to increasing amounts of food waste in industrial countries the separate collection for different treatment strategies is under discussion (Levis et al., 2010). Nevertheless, prevention of food waste should be given the highest priority. Regarding biogenic waste there is also a high potential in developing countries, especially for market waste, crop residues and manure. Two aspects suggest the use of these materials: the minimisation of the environmental risk due to uncontrolled emissions and resource recovery. This purpose is paralleled by adequate measures in terms of waste separation and collection. The separation of biogenic waste from municipal solid waste is not taken for granted in all European countries. In some cases biogenic waste is treated with municipal solid waste and only separated after the biological treatment. In Austria the source separation of biogenic materials was stipulated in the nineties by a corresponding ordinance (BMLFUW, 1992) in order to avoid diffuse contamination that can not be removed ex post. The Austrian Compost Ordinance (BMLFUW, 2001) provides a list of possible ingredients for composting in the first annex.

Composting processes are operated in open windrow or closed systems. The geometry of the windrow should allow efficient aeration by convection. Mechanical rotating supports air supply and the removal of volatile metabolic products which is very important during the most reactive phase of degradation. In closed systems forced aeration is necessary. The biological treatment consists of specific phases that are clearly distinguished from each other in well operated processes. The most obvious degradation with the highest transformation rate takes place in the intensive rotting phase that is characterised by increasing temperature due to exothermic reactions. Early metabolic products such as volatile fatty acids and ammonium are parameters that are usually applied to describe this stage of decomposition. The pH value allows a rough estimation. The early stage features low pH values of 5 to 6. In the mature compost the pH ranges from 7 to 8.5. Appropriate process operation in the first phase is relevant to reduce odour emissions by efficient air and water supply. Anaerobic conditions in the windrow lead to methane formation that should be avoided. Nevertheless, temporarily or locally limited aeration also supports humic substance formation that is improved by a moderate degradation to moieties of bio-molecules. Very strong aeration favours mineralisation. After the intensive rotting process metabolic activities slow down and change into the curing and the maturation phase. This stage is characterised by decreasing temperature, low respiration activity, a C/N ratio of about 12 and the oxidation of ammonium to nitrate. The stable stage is indicated by a nearly constant level of organic matter and total organic carbon contents respectively. The remaining organic matter consists of hardly degradable enriched substances, of organic substances stabilised by mineral compounds and of humic substances that are synthesised during composting. This process still takes place in the maturation phase. The continuous degradation process until the measured parameters reach a nearly constant level and indicate a stable product, is only achieved if the conditions for microbial activities are adhered to. A lack of water often gives the appearance of a "stable" state because it leads to a standstill of the microbial metabolism. Due to degradation of organic matter mineral compounds are enriched and show a relative increase. They mainly contribute to the stabilisation of the remaining organic matter fraction. Due to the portion in biogenic waste and the geological background carbonates and clay minerals play the most important role. Fig. 1 illustrates the development of the CO_2 concentration, temperature, respiration activity (RA_4) and humic acid contents in two composting processes in plant BC1 and plant BC2. Although process kinetics are individual according to the input material and operation conditions, principles of biological degradation are clearly visible. The respiration activities start at different levels and decrease continuously to a low value. The high temperature for several weeks guarantees favourable conditions regarding hygienic requirements. The CO_2 concentration in the windrow of plant BC2 is maintained for a longer time at a high level. The curves of humic acid formation are still increasing and indicate that the synthesis process has not yet been finished.

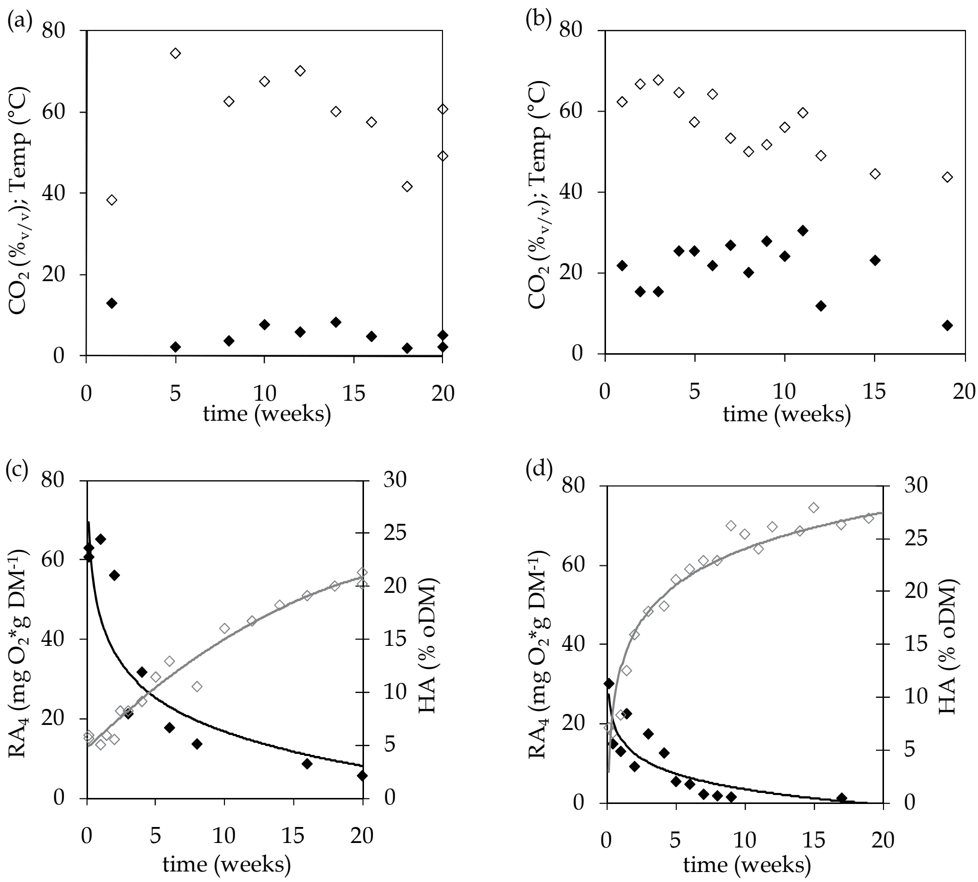

Fig. 1. Development of the parameters in the windrows: (a and b) CO_2 content (black dots) and temperature (grey symbols), (c and d) respiration activity (RA_4, black symbols), humic acids (HA, grey symbols); a and c = plant BC1, b and d = plant BC2

2.1.3 Influence of substrates on microbial communities

The composition of the organic waste mainly influences the turnover rates and the final product. Easily degradable ingredients such as sugars are quickly metabolised which can lead to strong acidification and cause the metabolism to stop. The addition of pH increasing agents such as calcite supports the regulation of the biological process. Fundamental requirements of the microbial metabolism affect the quality of the composting process (Schlegel, 1992). It mainly depends on the experience of the operator in the composting plant. Besides a lack of water and air, the pH values, the C/N ratio and the concentration of metabolic products are relevant parameters that can improve or reduce the microbial activity. If easily degradable materials are mineralised too fast hardly degradable substances are not attacked at all. This fact suggests that a well-balanced mixture of easily, middle and hardly degradable input materials is necessary to maintain the microbial activity, to regulate the velocity of transformation and to crack recalcitrant substances as well. Additionally it is a prerequisite for humic substance synthesis. A moderate progress of degradation provides the necessary molecule moieties and the opportunity to affect hardly degradable molecules such as lignin that is known to be a relevant compound of humic substances. Fig. 2 shows the respiration activity for 7 days (RA_7) and humic substance formation during two composting processes PI and PII, both operating biogenic waste, but considerably differing in the mixtures, especially in the fraction of medium degradable components such as grass clippings and leaves. The high microbial activity of process PI declined very fast due to an imbalanced mixture of easily and hardly degradable substances and humic acid contents remained at a low level. The microbial activity of process PII decreased more slowly. A constant increase of humic acid contents was observed. This fact underlines the assumption that moderate decrease of microbial activity supports humic acid formation in biowaste compost.

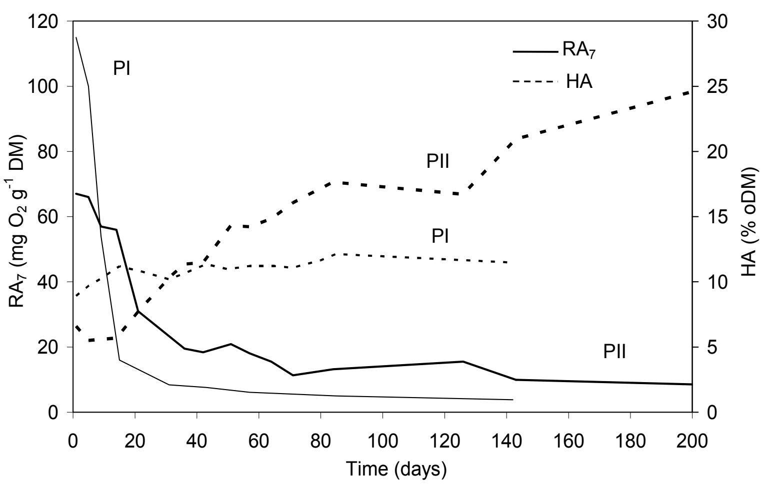

Fig. 2. Development of respiration activity (RA_7) and humic acid (HA) contents in two composting processes PI and PII (DM = dry matter; oDM = organic dry matter)

Due to the complexity of the material composition a large variety of microbial communities are involved in the metabolism of biogenic materials. A succession of different species is observed during the composting process (Franke-Whittle et al., 2009).

2.1.4 Process and product control by FT-IR spectroscopy and thermal analysis

The progress of composting processes can be monitored by means of near- and mid-infrared spectroscopy and thermal analysis. Both methods reveal the chemical changes during the biological degradation process by the characteristic spectral or thermal pattern. Several publications in the field of infrared spectroscopic investigations have focused on prediction models for parameters commonly used in waste management to describe compost quality (Michel et al., 2006; Böhm, 2009; Tandy et al., 2010). Fig. 3 illustrates a biowaste composting process using mid-infrared spectroscopy (Fig. 3a) and thermal analysis (Fig. 3b). It is evident that the progressing degradation process is reflected by both the spectral and the thermal pattern. The bands that are assigned to organic components tend to decrease corresponding to the biological degradation of the molecules. The transformation of organic substances causes some bands of metabolic products to emerge and disappear. The bands that can be attributed to inorganic compounds, e.g. carbonates and clay minerals, gain in height due to their relative increase. More detailed information on band assignment in waste materials were provided by Smidt and Schwanninger (2005) and Smidt and Meissl (2007). The aliphatic methylene bands labelled by arrows in Fig. 3a are relevant indicators of mineralisation that is revealed by decreasing band intensities. Fig. 3b illustrates the degradation of organic matter by the diminishing heat flow. After 14 days those substances primarily were degraded that contribute to the first exothermic peak at 320 °C. Besides the weaker intensities of both peaks after 120 days of composting a shift of the second exothermic peak by 10 degrees to higher temperature (490 °C) is observed. This behaviour is related to increasing stabilisation.

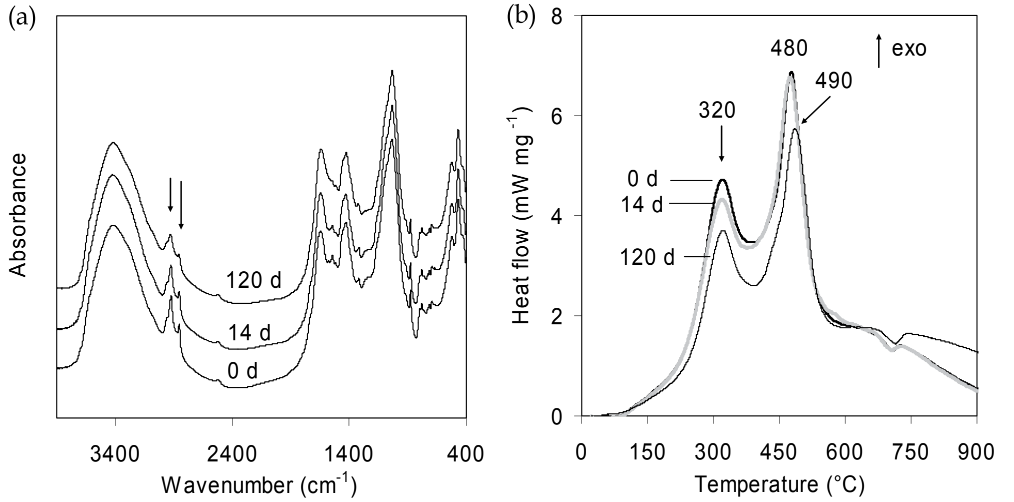

Fig. 3. Development of (a) infrared spectral and (b) thermal characteristics (heat flow profile) of biogenic waste during a composting process (selected stages: 0, 14 and 120 days)

The principal component analysis in Fig. 4 leads to the grouping of five composted materials due to spectral differences caused by the individual chemical composition. The materials of the Austrian biowaste composting processes Bio1, Bio2 and Bio3 can be distinguished as they differ in detail, but they are more similar to one another than the African biowaste (Bio4) composting process that is operated with locally available herbaceous materials. The difference of the sewage sludge compost (SSL) regarding the ingredients causes a large distance to biowaste composts in the scores plot of the principal component analysis (Fig. 4a). The biological degradation of different mixtures of biowaste and sewage sludge and biowaste and manure lead to a specific spectral pattern that is dominated by one of these components. In the biowaste/sewage sludge mixture the biogenic fraction is less resistant to microbial degradation than the anaerobically stabilised sewage sludge with a high portion of mineral compounds. By contrast, manure is faster degraded in the mixture biowaste/

manure and the spectral pattern becomes similar to the pure biowaste compost. The development with time is indicated by the arrows (Fig. 4b).

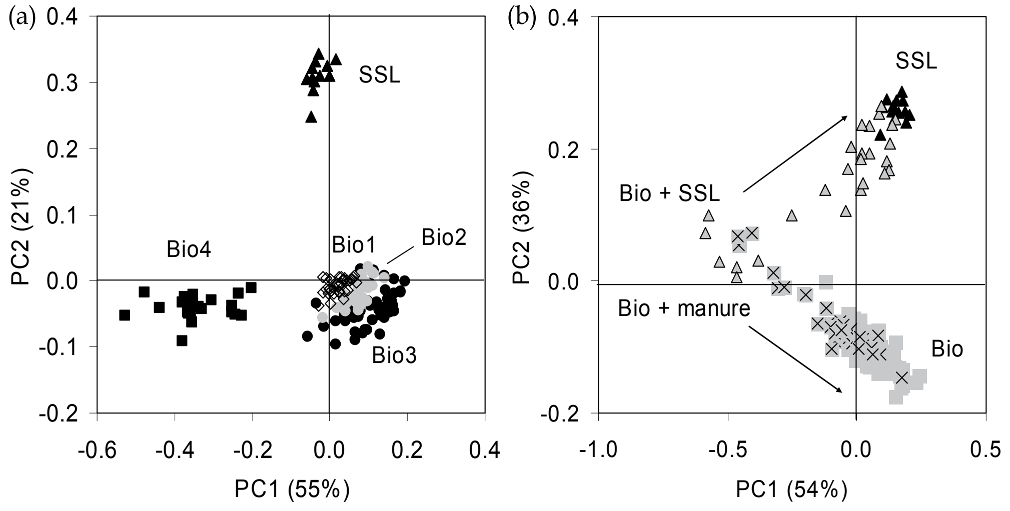

Fig. 4. (a) Principal component analysis of different composts based on their infrared spectral pattern (Bio1 – Bio4 = biowaste composts, SSL = sewage sludge compost); (b) different mixtures of biowaste/manure and biowaste/sewage sludge and their development during composting indicated by arrows

2.1.5 Quality criteria for composts

Due to practical reasons the description of compost organic matter is limited to quantitative determination of sum-parameters such as loss of ignition (LOI) and total organic carbon (TOC). Furthermore the nutrient content can be measured by the total nitrogen content (TN), phosphorous (P) and potassium (K) or mineralisation products such as ammonium nitrogen (NH_4-N) and nitrate nitrogen (NO_3-N). Information on stability can be given by the carbon to nitrogen ratio (C/N). A wide C/N ratio is typical for not degraded input materials. A C/N ratio of about 12 reflects stable compost matter. Stability also can be detected by other parameters such as degradable organic substance (AOS), "fractionation according to van Soest (1963)", biological (e.g. respiration activity, oxygen uptake rate) and plant compatibility tests. The parameters "degradable organic substance (AOS)" and the fractionation according to van Soest (1963)" focus on the specific degradability of organic matter fractions under different chemical conditions. Biological tests describe the behaviour of organic matter and therefore provide indirect information on reactivity and stability. The mentioned parameters do not provide any information on organic matter quality. Information on organic matter quality is provided by humic substance determination. More detailed insight into the chemical composition is available by sophisticated analytical tools. Fourier Transform infrared (FT-IR) spectroscopy, nuclear magnetic resonance (NMR) spectroscopy, eco-toxicity tests and thermal analysis are currently applied in research, and apart from NMR spectroscopy these tools are intended for future practical application.
The mentioned parameters and analytical tools, the information they provide and related references are compiled in table 1 and table 2 (Böhm, 2009). The parameters and methods

mainly focus on process control and the determination of maturity. Therefore they are related to organic matter and the mineralisation products.

Parameter	Information on	Related references
Loss of ignition (LOI)	Quantity of organic matter	(Austrian Standard Institute, 1993)
Total organic carbon (TOC)	Quantity of organic matter	(Austrian Standard Institute, 1993)
Kjeldahl nitrogen (Nkjel)	(potential) Nutrient content	(Austrian Standard Institute, 1993)
Mineralisation products (NH_4-N, NO_3-N)	Nutrient content	(Austrian Standard Institute, 1993; Haug, 1993)
Low molecular weight carboxylic acids (C-2 to C-5)	Reactivity, Odour index	(Haug, 1993; Binner & Nöhbauer, 1994; Lechner & Binner, 1995)
C/N	Stability	(Austrian Standard Institute, 1993; Haug, 1993; Barberis & Nappi, 1996; Ouatmane et al., 2000)
Degradable organic substance (AOS)	Degradation behaviour	(Austrian Standard Institute, 1993)
Fractionation of organic matter according to van Soest	Degradation behaviour	(van Soest, 1963)
Biological tests e.g. oxygen uptake rate, respiration activity, enzymatic tests	Degradation behaviour, Stability	(Haug, 1993; Barberis & Nappi, 1996; Lasaridi & Stentiford, 1996; Lasaridi & Stentiford, 1998; Chica et al., 2003; Adani et al., 2004; Barrena GoÌmez et al., 2006)
Plant compatibility	Stability, Maturity	(Zucconi et al., 1981a; Zucconi et al., 1981b; Austrian Standard Institute, 1993; Haug, 1993; Commission of the European Communities, 2008)
Humic substances	Quality of organic matter	(Adani et al., 1995; Barberis & Nappi, 1996; Senesi & Brunetti, 1996; Ouatmane et al., 2000; Tomati et al., 2000; Zaccheo et al., 2002; Smidt & Lechner, 2005; Meissl et al., 2007)

Table 1. Parameters used for the characterisation of compost organic matter, the information they provide and related references (adapted from Böhm, 2009)

Besides maturity the content of toxic compounds plays a crucial role. Especially in countries where the application of pesticides in yards and gardens is very common, the contamination of biogenic input materials with organic pollutants can be relevant. Saito et al. (2010) reported on the concentration of clopyralid in composts. With regard to inorganic pollutants heavy metals are in the focus of interest. They remain in the cycle and are accumulated with

repeated compost application. Therefore their content in the compost is limited. The classification of compost quality according to the Austrian Compost Ordinance (BMLFUW, 2001) is based on limit values of several heavy metal contents.
Besides the standard quality parameters that mainly focus on the reduction of negative impacts, additional quality criteria are useful that emphasise the positive effects of composts and underline the value as marketable products. Nutrients and their availability (Gil et al., 2011), the content of humic substances (Tan, 2003; Böhm et al., 2010) and phytosanitary effects (Pane et al., 2011) might be appropriate parameters to meet this purpose.

Parameter	Information on	Related references
Spectroscopic methods Infrared spectroscopy (IR) Nuclear magnetic resonance (NMR)	Information on a molecular level. Identification of specific molecules and molecule groups Stability, Quality	FT-IR spectroscopy: (Ouatmane et al., 2000; Chen, 2003; Smidt et al., 2005; Smidt & Schwanninger, 2005) NMR spectroscopy: (Kögel-Knabner, 2000; Zaccheo et al., 2002; Chen, 2003; Tang et al., 2006)
Ecotoxicity tests	Indirect information on toxic compounds and therefore on compost quality	(Barberis & Nappi, 1996; Kapanen & Itävaara, 2001; Alvarenga et al., 2007)
Thermal analysis Thermogravimetry (TG) and Differential scanning calorimetry (DSC) Pyrolysis field ionisation mass spectrometry (Py-FIMS) Pyrolysis gas chromatography mass spectrometry (Py-GC/MS)	Stability (TG and DSC) Information on the molecular level with coupled MS	(Dell'Abate et al., 2000; Ouatmane et al., 2000; Otero et al., 2002; Melis & Castaldi, 2004; Dignac et al., 2005; Smidt et al., 2005; Smidt & Lechner, 2005; Franke et al., 2007)

Table 2. Analytical tools used for the characterisation of compost organic matter, the information they provide and related references (adapted from Böhm, 2009)

Compost teas that are fermented aqueous extracts from composts are suggested by several authors as an alternative to mineral fertilisers and pesticides (Koné et al., 2010; Naidu et al., 2010).

2.2 Anaerobic digestion

Anaerobic digestion of biogenic materials is a booming technology as it combines organic matter stabilisation and energy recovery. The high interest in this technology is paralleled by the question how to handle the increasing amounts of digestates. Due to a limited retention time in the reactor digestates still feature a considerable reactivity. Therefore open systems for their storage such as lagoons, can lead to uncontrolled emissions of methane or

N_2O. The quantification of these relevant greenhouse gas compounds from these sources has not been done yet. The useful application of digestates becomes an important question in terms of available areas and transport distances. Digestates are directly used in agriculture or subjected to further treatment such as composting. Eco-balances are necessary to oppose the advantages to the disadvantages and to evaluate the benefits. Increasing pH values during the subsequent composting process lead to ammonia losses. Their determination is an additional question to be answered (Whelan et al., 2010). Nitrogen recovery can be provided by ammonia stripping (De la Rubia et al., 2010; Zhang & Jahng, 2010).

2.2.1 Adequate ingredients and process operation

Basically most of the biogenic waste materials are appropriate for anaerobic digestion and only restricted by natural limitations of biodegradability. Lignin for instance is not degradable under anaerobic conditions. Steam explosion of lignocellulosics is a kind of pre-treatment of wooden materials in order to remove cellulosic compounds from the composite lignin and to make them available to microorganisms. Kitchen and market waste from the separate collection, mainly consisting of easily degradable components also serve as input materials for composting processes. By contrast, leftovers originating from public institutions, hospitals, hotels and schools are appropriate ingredients for anaerobic processes and do not compete with other ways of utilisation. Besides organic wastes from urban areas agricultural wastes such as liquid and solid manure and crop residues are processed locally in biogas plants. Industrial waste from the food industry and biotechnological processes, slaughterhouse waste and sewage sludge complete the wide range of organic substances that are processed in anaerobic digesters (Lee et al., 2010). The anaerobic treatment of sewage sludge is more a part of the waste water treatment. Anaerobically stabilised sludge that undergoes a composting process, comes within the limits of waste management. In Austria slaughterhouse waste is subject to restrictions in order to guarantee hygienic standards (Europäische Union, 2009).

Anaerobic digestion is usually carried out under mesophilic (~37°C) or thermophilic (~55°C) conditions (Madigan et al., 2003). Depending on the water content "wet" (5-25% dry matter) and "dry" (>25-55% dry matter) processes are distinguished. The water content also determines to a certain degree the fate of digestates. Very low contents of dry matter suggest an immediate application on fields by irrigation. If an additional composting process is planned, solid residues are in general separated by centrifugation or by a filter press. Anaerobic digestion at relatively low water contents allows a subsequent composting process. Whereas methanogenesis in a liquid process takes place in a temporal sequence, the dry process is dominated by the spatial sequence, depending on the motion of the bulk along the reactor. Anaerobic digestion plays a certain role as pre-treatment of municipal solid waste in several countries (Fdez.-Güelfo et al., 2011; Lesteur et al., 2011) in order to yield biogas before aerobic treatment and final disposal. Improvement of biogas production is a main target and many investigations focus on this issue. The organic fraction of municipal solid waste underwent a dry thermophilic anaerobic digestion process to find out the optimum solid retention time in the reactor regarding the gas production (Fdez.-Güelfo et al., 2011). Co-digestion of press water from municipal solid waste and food waste could improve the gas yield according to Nayono et al. (2010). Besides the substrate process conditions play an important role in terms of gas yields.

Different procedures and technologies are suggested to upgrade the resulting biogas regarding the purity degree of methane (Dubois & Thomas, 2010; Poloncarzova et al., 2011).

2.2.2 Microbial communities in anaerobic processes

Molecular identification of microbial communities depending on substrates and process operation and their dynamics during anaerobic digestion were reported by several authors. Organic waste and household waste were used as substrates in these studies (Ye et al., 2007; Hoffmann et al., 2008; Montero et al., 2008; Cardinali-Rezende et al., 2009; Sasaki et al., 2011). Shin et al (2010) reported on characteristic microbial species that dominate specific phases of a food waste-recycling wastewater digestion process and therefore provide information on the performance of the reactor. Hydrolysis efficiency in a similar substrate and the related microbial communities were investigated by Kim et al. (2010). Wagner et al (2011) reported on diverse fatty acids such as acetic, propionic and butyric acid that inhibited methanogenesis coupled with an increase of hydrogen. Abouelenien et al. (2010) could improve the methane production by removal of ammonia that had a negative impact on methanogenesis. By contrast, elevated ammonia contents did not inhibit methanogenesis in a co-digestion process of dairy and poultry manure (Zhang et al., 2011). Although basic mechanisms of the anaerobic metabolism are well-known it should be emphasised that the results obtained are divergent according to the wide range of different experiments regarding individual feeding materials and process conditions. The identification of various microbial communities reflects the complexity of interactions in these processes.

2.2.3 Residues from anaerobic digestion

Chemical characteristics of digestates are mainly influenced by the input material, process conditions and the retention time in the reactor. High salt concentrations caused by leftovers do not pose a problem for the subsequent composting process as they are removed with the waste water. By contrast, heavy metals primarily remain in the solid residue. As mentioned above the water content usually determines the further treatment or immediate application on fields. Quality criteria of the liquid residue that is directly applied on the field are mainly determined by the quality of the input material. The nitrogen content is the limiting compound according to the Water Act (Wasserrechtsgesetz BGBl. Nr. 215/1959, in der Fassung BGBl. I Nr. 142/2000) that regulates the output quantity. Composting is a suitable measure to stabilise digestates and to produce a valuable soil conditioner. Disaggregation of the wet, cloggy and tight material is a main issue to ensure the porosity and the efficient air supply. Wood particles and yard waste are appropriate bulking agents for this purpose.

2.2.4 Process control by FT-IR spectroscopy and thermal analysis

Parameters usally applied in waste management such as the loss on ignition, total organic carbon contents and total nitrogen describe degradation and changes of organic matter during anaerobic digestion and composting. The reactivity of the material is measured using biological tests. The oxygen uptake during a period of 4 days reflects the current microbial activity (RA_4), whereas the gas sum (GS_{21}) indicates the gas forming potential under anaerobic conditions during a period of 21 days. Compared to time-consuming biological tests modern analytical tools provide fast information on reactivity and material characteristics. Near infrared spectroscopy was used by Lesteur et al (2011) to predict the biochemical methane potential of municipal solid waste. Fig. 5a demonstrates the development of the mid-infrared spectral pattern from the reactor feeding mixture (FM) to the digestate (D) and the composted digestate (DC). The samples originate from the Viennese biogas plant that processes 17,000 tons a year of biogenic waste from the separate collection, market waste and leftovers. The thermograms in Fig. 5b illustrate the mass losses of these samples. The degradation of organic matter becomes evident by decreasing mass losses.

The spectrum of the feeding mixture features a variety of distinct bands in the fingerprint region (1800-800 cm^{-1}) and high intensities of the aliphatic methylene bands. The breakdown of biomolecules due to degradation is paralleled by their decrease. Distinct bands in waste materials indicate a variety of not degraded substances. With increasing degradation bands tend to broaden in the complex waste matrix. The band at 1740 cm^{-1} can be attributed to the C=O vibration of carboxylic acids, esters, aldehydes and ketones, indicating an early stage of degradation. The bands at 1640, 1540 and 1240 cm^{-1} represent different vibrations of amides (C=O, N-H, C-N). Typical absorption bands of carboxylates (C=O) and alkenes (C=C) are also found at 1640 cm^{-1}. More detailed information on band assignment is provided by Smidt and Schwanninger (2005) and Smidt and Meissl (2007). Digestates are still reactive after a retention time of 21 days in the reactor. The remaining gas sum over a period of 21 days (GS_{21}) was 80 to 120 L per kg dry matter. The total nitrogen content was found to be between 4 and 5% referring to dry matter (DM). The total nitrogen content in digestate composts was about 2.5% (DM). The nitrogen content is higher than in biowaste composts that feature 1-2% of total nitrogen (DM). Nevertheless, the losses during composting are considerable and need more attention in the future. Apart from the losses of this nutrient compound the volatilisation of ammonia that is formed at higher pH-values leads to odour nuisance in open windrow systems and represents one of the relevant problems in this type of composting plants.

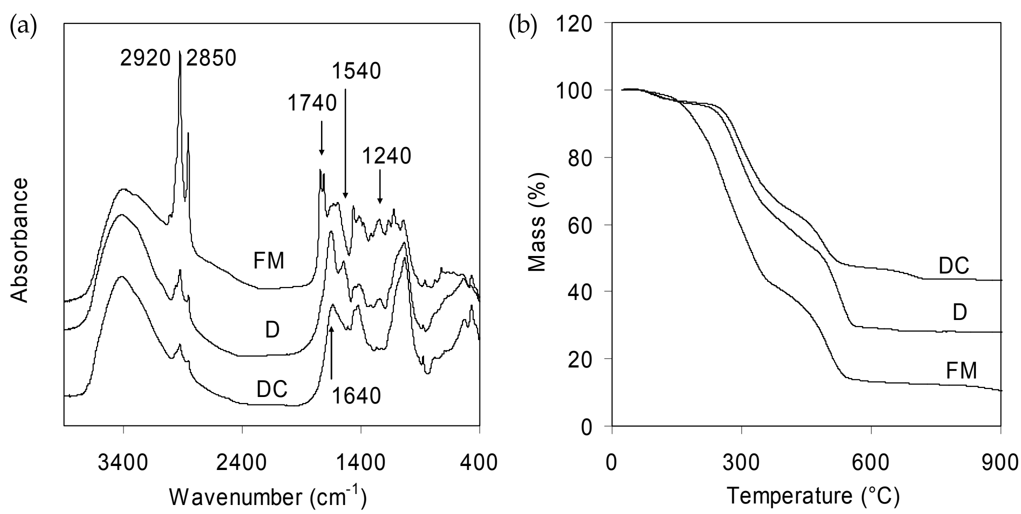

Fig. 5. Development of (a) the spectral and (b) the thermal (mass loss) pattern during anaerobic digestion and subsequent composting of digestates (FM = feeding mixture for the reactor, D = digestate, DC = digestate compost)

Depending on the input materials digestates keep a specific pattern. A principal component analysis based on FT-IR spectra reveals the similarity of residues that originate from thermophilic processes with a 2 to 3 week-retention time in the reactor (Fig. 6a). Three groups of digestates according to the input materials can be distinguished: manure, biowaste comprising yard waste, fruits and vegetables from households and markets and leftovers that had been mixed with different amounts of glycerol. The 1st principal component explains 85% of the variance, the 2nd one 7%. The loading plots indicate the spectral regions that are responsible for the discrimination of the materials: the aliphatic methylene bands at 2920 and 2850 cm^{-1}, and nitrogen containing compounds such as amides at 1640 and 1540 cm^{-1} and nitrate at 1384 cm^{-1} (Fig. 6b).

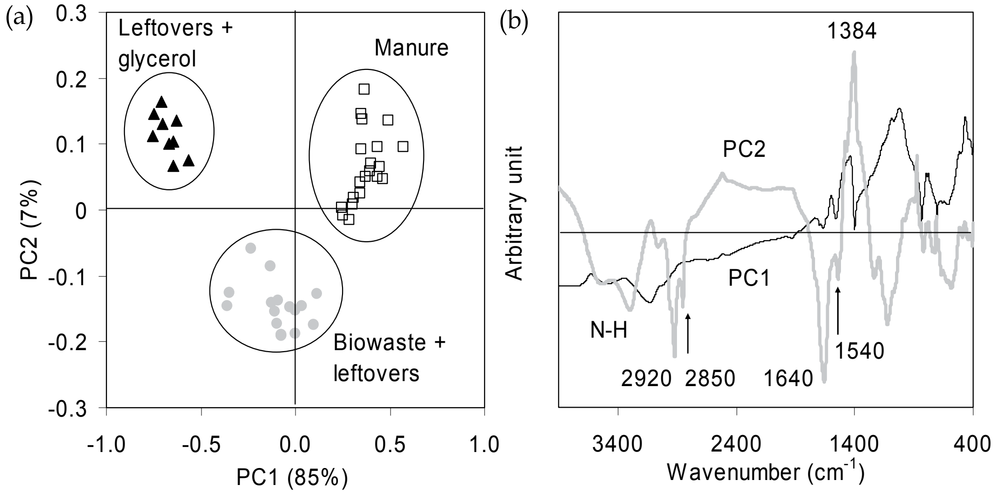

Fig. 6. (a) Principal component analysis based on infrared spectra of digestates from different input materials that underwent thermophilic processes; (b) corresponding loadings plot of the first two principal components

3. Biological treatment of municipal solid waste for safe final disposal

Biological processes always deal with both aspects: resource recovery and the avoidance of negative emissions. The history of waste management started with harmful emissions. Waste was disposed in open dumps and was used to level off depressions in the landscape or to fill and dry wet hollows. This strategy has caused severe problems with increasing amounts of waste. The dumped waste was degraded anaerobically, metabolic products of early degradation stages were leached and washed out to the groundwater. Gaseous emissions leaked from the dumps to the top and into the atmosphere or migrated into nearby cellars which can cause an explosion if the critical mixture of methane with air is reached. These environmental problems have led to regulations about the technical demands on landfill sites. The idea was to prevent the emissions by closing the landfills with dense layers at the bottom and on the top to cut them from the environment. Actually the degradation processes continued and the emissions were sealed and preserved, but not prevented. It can be assumed that the life time of the technical barriers is over after some decades. The emissions, leachate at the bottom and landfill gas on the top, become relevant as soon as the density of the layers fails. This fact has promoted the latest changes in European regulations. The stabilisation of waste organic matter prior to landfilling was proclaimed and with regard to the biological treatment the natural stabilisation processes served as a paradigm.

3.1 Mechanical-biological treatment (MBT) of municipal solid waste

Besides incineration mechanical-biological treatment is one option to stabilise municipal solid waste prior to final disposal. The mechanical-biological treatment of waste combines material recovery and stabilisation before landfilling. Big particles, especially plastics with a high calorific value, are separated by the mechanical treatment and used as refused derived fuels. The residual material features a relatively low calorific value, a high water content and a high biological reactivity. The calorific value is mainly influenced by the content of organic matter. The biological treatment abates all three parameters. Organic matter is degraded by microbes which leads to gaseous and liquid emissions. Due to the exothermic aerobic biological process the temperature rises. Water evaporates due to the generated heat and the material tends to run dry. The decrease of organic matter that is paralleled by the relative increase of inorganic compounds causes the calorific value to decrease. The degradation process is dominated by mineralisation. Depending on the input material humification takes place to a certain extent. Mineral components contribute to organic matter stabilisation. In practice MBT processes vary in many details. Apart from stabilisation of the output material for landfilling the biological process can focus on the evaporation of water to produce dry material for incineration. Another modification of the process provides anaerobic digestion prior to aerobic stabilisation in order to yield biogas in addition. Most of the MBT plants are situated in Germany and Austria. In France the biogenic fraction is not source separated and thus treated together with municipal solid waste. The output material is used as waste compost and applied on soils. In Germany and Austria this procedure is prohibited by national rules. In this section the MBT technology is described as it is implemented in Germany and Austria. The system configuration of the plants is described in Table 3.

plant	input material	system	mesh size/ treatment
A	MSW	4 w cs, 8-14 w rp	80 mm cs, 60 mm rp, 45 mm lf
B	MSW, SS	2 w cs, 6-8 w rp	80 mm cs, rp, 25 mm lf
C	MSW	4 w cs, 8 w rp	80 mm cs, rp, 25 mm lf
D	MSW	3-4 w cs, 7-9 w rp	160 mm cs, 20 mm rp, lf
E	MSW	4 w cs, 8 w rp	80 mm cs, rp, 40 mm lf
F	MSW	5 w cs, 10-30 w rp	25 mm cs, rp, lf
G	MSW	60-80 w cs+rp	80 mm
H	MSW	30 w cs+rp	25 mm
J	MSW, ISW	20 w cs+rp	70 mm cs+rp, 25 mm lf
K	MSW	4 w cs, 10 w rp	70 mm cs, rp, 30 mm lf
L	MSW	4 w cs, 20 w rp	50 mm cs, rp, lf
M	MSW, SS, BW	10 w cs, 40-60 w rp	60 mm cs, rp, 12 mm rp, 9 mm cp
N	MSW	4 w cs, 12 w rp	80 mm cs, 10 mm rp, lf
O	MSW	3 w bd	40 mm bd, ~25 mm lf
P	MSW, BW	9 w + 6 w rp	not sieved, 20 mm rp, 10 mm cp
R	MSW	6-8 w bd	100 mm bd
S	MSW	1-2 w bd	80 mm bd

Table 3. Austrian MBT plants, input materials and systems applied (MSW: municipal solid waste; SS: sewage sludge; BW: biowaste; ISW: industrial solid waste; cp: compost, cs: closed system; bd: biological drying; rp: ripening phase; lf: landfilled; w = week)

This table displays the diversity of the Austrian mechanical biological treatment processes regarding input materials, mesh size and the duration of rotting and ripening phases in open or closed systems (adapted from Tintner et al., 2010). In Germany about 50 plants are in operation, in Austria 17. Two Austrian plants produce exclusively refuse derived fuels.
Anaerobic digestion prior to the aerobic treatment is currently not performed in Austrian MBT plants.

3.2 Stabilisation of waste organic matter

The aerobic biological stabilisation process comprises in general two main phases. The first intensive rotting phase takes place in a closed box with forced aeration. The ripening phase proceeds in open windrows, sometimes covered with membranes. The respiration activity that reflects the reactivity of the material summarises the oxygen uptake (mg O_2 g^{-1} DM) by the microbial community over a period of four days. The respiration activity of input and output, 4-week-old and already landfilled material originating from different Austrian plants was measured. In two plants also waste compost was produced which has ceased in the meantime. Results for mean values and the confidence intervals are given in Table 4 .

Respiration activity (mg O_2 * ·g^{-1} DM)			
	mean	c_l	c_u
Input material $_{n=34}$	44.4	38.3	50.4
After 4 weeks $_{n=19}$	24.1	15.8	32.4
Output material $_{n=53}$	6.9	5.3	8.5
Waste compost $_{n=9}$	7.8	2.6	13.0
Landfilled material $_{n=13}$	6.4	2.7	10.1

Table 4. Respiration activity over four days in mg O_2*g^{-1} DM; c_l: lower bound of confidence interval, c_u: upper bound of confidence interval, $\alpha = 0.05$

Depending on the system process kinetics can considerably differ regarding the decrease of reactivity. Fig. 7 presents the degradation of organic matter in three different plants (plants D, O, and P according to Table 3). The input material in plant P consists of municipal solid waste and biowaste that had not been separated. This mixture results in a highly reactive input material compared to the other plants. Plant O provides a wind cyclone for the separation of the heavy fraction after a three-week treatment. Plant D represents the classical MBT-type with a three-week intensive rotting phase in a closed system and a seven to nine-week ripening phase in an open windrow system (Tintner et al., 2010). The biological degradation of MBT materials corresponds to the biological degradation in composting processes.
In Fig. 8 the degradation processes in plants M and H are presented in more detail. The CO_2 concentration and the temperature in the windrows are compared to the water content and the respiration activity of the material. In both plants the respiration activity decreases continuously according to organic matter mineralisation. The CO_2 concentration depends on the system configuration. In the closed system of plant M the material is aerated actively for 10 weeks. Thereby the oxygen supply is ensured most of the time. In plant H no forced aeration is provided. The CO_2 content increases up to 60 %. However, these temporarily anaerobic conditions in some sections do not inhibit the biological degradation as the material is turned regularly. The efficient aerobic degradation is verified by the high temperature. It is remarkable that the temperature of the windrow remained at a high level

for a long time. The high temperature supports sanitation of the material which plays a secondary role for MBT output that is landfilled, compared to compost. Although the respiration activity decreased considerably further microbial activities took place, indicated by the constant high level of CO_2 contents in the windrow. Inefficient turning might have been the reason for the CO_2 contents and the high temperature.

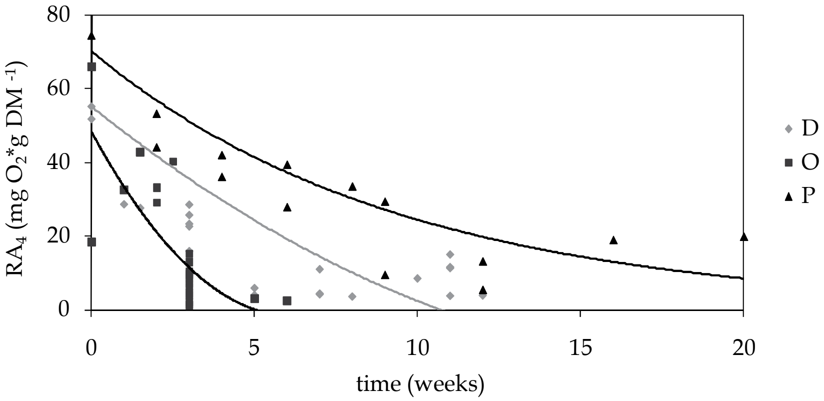

Fig. 7. Decrease of the respiration activity (RA_4) in three different MBT-plants with different operation systems

The data reflect process kinetics by the specific pattern of organic matter degradation during the biological treatment of MBT materials. The principles of the metabolism are the same as in composting processes. However, the individual mixtures of input materials and system configuration strongly influence the transformation rate. The period of time that is necessary to comply with the limit values of the Landfill Ordinance (BMLFUW, 2008) is a main factor for successful process operation. It should be emphasised that water and air supply play a key role in this context and the retardation of organic matter degradation can in general be attributed to a deficiency of air and water. A homogenous distribution of air in the windrow and the removal of metabolic products is only guaranteed by regular mechanical turning.

3.3 Landfilling

When the legal requirements are reached the treated output material is landfilled. The most relevant parameters are the respiration activity with limit values of 7 mg*g DM^{-1} in Austria and 5 mg*kg DM^{-1} in Germany and the gas generation sum that provides information on the behaviour of waste materials under anaerobic conditions. The determination of the gas generation sum is obligatory in Austria and facultative in Germany. In both countries the limit value is 20 NL*kg DM^{-1}.

Landfilling is usually performed in layers of about 20 to 30 cm. The material is rolled by a compactor. In some cases a 40-centimetre drainage layer of gravel is integrated every 2 metres between the waste material. The degree of compaction depends on the water content. At the end of the biological process the material is often dried out. This advantage for the sieving process counteracts the optimal compaction because the water content is lower than the necessary proctor water content. However, a satisfactory coefficient of permeability of about 10^{-8} m/s is usually achieved. The efficient compaction can be one of the main reasons why further degradation processes in the landfill are reduced to a minimum. As indicated in

Table 4 the reactivity (mean value) of the landfilled material and of the MBT-output material is similar.

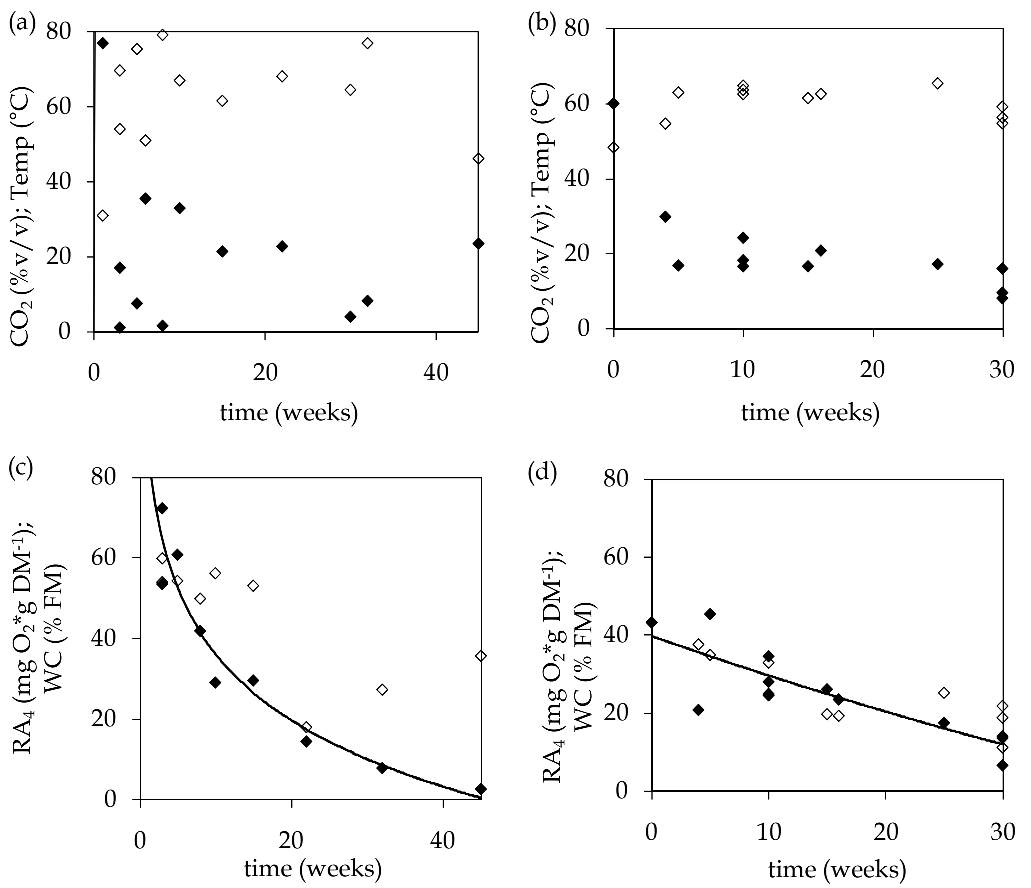

Fig. 8. (a and b) Development of the parameters in the windrow: CO_2 content (black symbol) and temperature (circle), (c and d) respiration activity (RA_4, black symbol), water content (WC, circle); a and c = plant M, b and d = plant H

In six different MBT plants one to four year-old landfilled materials were compared to the typical output material of these plants after the biological treatment. The comparison of the respiration activity confirmed that no significant degradation took place in the landfill.

Biological degradation after landfilling is minimised and the remaining organic matter is quite stable which is the main target of the pre-treatment of municipal solid waste. However, low methane emissions can be expected. These emissions are mitigated by means of methane oxidation layers where methanotrophic bacteria transform methane into CO_2 (Jäckel et al., 2005; Nikiema et al., 2005). Several publications have focused on the identification of the involved methanotrophs (Gebert et al., 2004; Stralis-Pavese et al., 2006). Regarding the discussion about landfills as carbon sinks the question arises, how much carbon can finally be stored in MBT landfills. The remaining carbon content in MBT landfills can be considered as a stable pool, taken out of the fast carbon cycle. The mean content of organic carbon of the landfilled materials was 15.6 % DM at a 95 %-confidence interval from 13.3 to 17.8 % DM. The fitting model of the final degradation phase is a topic of current research.

3.4 Process control by FT-IR spectroscopy and thermal analysis

Besides the time consuming conventional approaches for the determination of the biological reactivity in MBT materials FT-IR spectroscopy was proven to be an adequate alternative. The prediction model for the respiration activity (RA_4) and the gas generation sum (GS_{21}) presented in Böhm et al. (2010) are based on all degradation stages and types of MBT materials existing in Austria.
The second relevant parameter to be measured prior to landfilling is the calorific value. This parameter is usually determined by means of the bomb calorimeter. An alternative method of determination is thermal analysis. The prediction model described by Smidt et al. (2010) is also based on all stages and types of MBT materials existing in Austria.

4. Abandoned landfills from the past and related problems

Although microbial processes lead to mineralisation of waste organic matter and finally to the stabilisation by mineralisation, interactions with mineral compounds or humification, degradation is paralleled by harmful emissions if it is not managed under controlled conditions. The amount and the particular composition of municipal solid waste lead to the imbalance of the system. Careless disposal of municipal solid waste and industrial waste in the past has caused considerable problems in the environment. Due to anaerobic degradation of waste organic matter groundwater and soils were contaminated. The discussions on climate change have attracted much attention on relevant greenhouse gas emissions in this context, especially on methane. Emissions of nitrous oxide from landfills have not been quantified yet. This awareness has led to adequate measures in waste management. As mentioned in the previous section the treatment of municipal solid waste before final disposal is a legal demand in order to have biological processes taken place under controlled conditions.

4.1 Risk assessment and remediation measures of contaminated sites

Despite national rules risk assessment of old landfills and dumps is still a current topic. In countries without an adequate legal frame for waste disposal it will be for a long time. Landfill assessment usually comprises the measurement of gaseous emissions on the surface. Due to inhibiting effects such as drought that prevent mineralisation, the investigation of the solid material is suggested as it reveals the potential of future emissions. Basically the analytical methods FT-IR spectroscopy and thermal analysis are appropriate tools to assess the reactivity of old landfills and dumps (Tesar et al., 2007; Smidt et al., 2011). Biological tests using different organisms provide information on eco-toxicity. The advantage of this approach is the overall view on the effect not on the identification of several selected toxic compounds (Wilke et al., 2008). This procedure is less expensive and in many cases, especially in old landfills containing municipal solid waste, sufficient. Nevertheless, until now the identification and quantification of single organic pollutants and heavy metals is the common approach.
Depending on the degree of contamination specific measures of remediation are required. Excavation of waste materials is the most extreme and expensive way of sanitation. The

presence of hazardous pollutants can necessitate such procedures. In many cases the reactivity of organic matter is the prevalent problem and mitigation of methane by a methane oxidation layer is an adequate measure. In-situ aeration is an additional approach to avoid methane emissions. Due to the forced aeration of the waste matrix in the landfill aerobic conditions replace anaerobic ones. They accelerate and favour the biological degradation of organic matter to CO_2.

4.2 Re-use and land restoration

As a consequence of the new strategy of waste stabilisation prior to landfilling the possibility of re-use and land restoration for after use becomes evident. Especially the demand for space for the production of renewable energy crops has promoted the awareness of a more economical and considerate exploitation of land. The typical landfill emissions in the past restricted the potential for many after use concepts. Landfill gas minimises the feasibility for agricultural purposes. Therefore most of the old landfill sites are not in use at all. The alternatives for after use concepts range from highly technical facilities or leisure parks to natural conservation areas. Even when the production of food on landfill sites is not taken into account agricultural use for the production of energy crops (maize, wheat, elephant grass, short rotation coppice) has a great potential (Tintner et al., 2009). There are some constraints such as climatic conditions, soil properties, soil depth, compaction, water availability and drought, waterlogging, aeration, and the nutrient status. Provided that no or just negligible landfill gas emissions are present in the root zone, careful site management including a correct soil placement and handling, soil amelioration, irrigation respectively drainage depending on precipitation, fertilisation, choice of adequate species, can accomplish the necessary environmental conditions (Nixon et al., 2001). Remediation of the sites is just a prerequisite for a successful land use management.

5. Conclusion

The biological treatment of organic waste materials is state of the art in Austria. Two main strategies are in the focus of interest: stabilisation of organic matter for safe waste disposal or landfill remediation and production of biogas and composts. The biological treatment of waste matter takes place according to the principles of the microbial metabolic pathways. The knowledge of fundamental requirements determines the quality of process operation. Water and air supply is a key factor in aerobic processes and mainly influences the progress of degradation besides the pH value and the nutrient balance. Water and air supply only depend on process operation, the nutrient balance is preset by the incoming waste material mixture. In small treatment plants it can be influenced marginally. The pH value is rather a result of input materials and process operation. Anaerobic digestion for biogas production requires more technical control to maintain a constant gas yield. Microbial processes always take place. It is a matter of anthropogenic activities to avoid the negative impact on the environment, but to use the potential of microbial processes.

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Development of On-Farm Anaerobic Digestion

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1. Introduction

Although humankind has always relied on generating energy from biomass in some form (e.g. firewood), it has only recently been re-conceptualised as 'bioenergy'. This is possibly because it was seen as an anachronism in the developed world for most of the last century (Plieninger et al., 2006). About 80% of the world's energy supply is currently derived from fossil fuels, but of the renewable energy sources, biomass is still by far the most important with between 10 to 15% of demand (or about 40-50 EJ per year).

'Biomass' is biological material derived from living, or recently living organisms such as forest residues (e.g. dead trees, branches and tree stumps), green wastes and wood chips. A broader definition of biomass also includes biodegradable wastes and residues from industrial, municipal and agricultural production. It excludes organic material which has been transformed by geological processes into substances such as coal or petroleum. In industrialised countries biomass contributes some 3–13% of total energy supply, but in developing countries this proportion is much higher (up to 50% or higher in some cases).

The recent scientific interest in bioenergy can be traced through three main stages (Leible & Kälber, 2005, cited in Plieninger et al., 2006): the first stage of discussion started with the 1973 oil crisis and the publication of the Club of Rome's report on 'The Limits to Growth'. Along with Rachel Carlson's 'Silent Spring', the Limits to Growth report was an iconic marker of the environmental movement's emergence and a precursor to the concept of sustainable development. The second stage of interest in bioenergy began in the 1980s in Europe as a result of agricultural overproduction and the need to diversify farm income. Triggered by increasing concern over climate change, a third stage started at the end of the 1980s, and continues to this day.

In the early years of expansion in renewable energy technologies, bioenergy was considered technologically underdeveloped compared with wind energy and photovoltaics. Now biomass has proved to be equivalent and in some aspects even superior to other renewable energy carriers. Technological progress facilitates the use of almost all kinds of biomass today – far more than the original firewood use (Plieninger et al., 2006). Biomass has the largest unexploited energy potential among all renewable energy carriers and can be used for the complete spectrum of energy demand – from heat to process energy and liquid fuel, to electricity.

Direct combustion is responsible for over 90% of current secondary energy production from biomass. Biomass combustion is one of the fastest ways to replace large amounts of fossil fuel based electricity with renewable energy sources. Biomass fuels like wood pellets and palm oil can be co-fired with coal or fuel oil in existing power plants. In a number of European countries, heat generated by biomass provides up to 50% of the required heat energy. Wood pellets, have become one of the most important fuels for both private and commercial use. In 2008, approximately 8.6 m tonnes of wood pellets were consumed in Europe (excluding Russia) with a worldwide total of 11.8 m tonnes (German Federal Ministry of Agriculture and Technology, 2009). In Germany, the number of wood pellet heating systems installed in private homes has increased from around 80,000 in 2007 to approximately 105,000 in 2008.

Anaerobic digestion (AD) currently plays a small, but steadily growing role in the renewable energy mix in many countries. AD is the process by which organic materials are biologically treated in the absence of oxygen by naturally occurring bacteria to produce 'biogas' which is a mixture of methane (CH_4) (40-70%) and carbon dioxide (CO_2) (30-60%) plus traces of other gases such as hydrogen, hydrogen sulphide and ammonia. The process also produces potentially useful by-products in the form of a liquid or solid 'digestate'.

It is widely used around the world for sewage sludge treatment and stabilisation where energy recovery has often been considered as a by-product rather than as a principal objective of the process. However, in several European countries anaerobic digestion has become a well established energy resource and an important new farm enterprise, especially now that energy crops are increasingly being used.

2. Historical development of anaerobic digestion

Anecdotal evidence indicates that biogas was used for heating bath water in Assyria during the 10th century BC and in Persia during the 16th century BC (Wellinger, 2007). The formation of gas during the decomposition of organic material was first described by Robert Boyle and Denis Papin in 1682 (Braun, 2007) but it was 1804 by the time John Dalton described the chemical formula for methane.

The first anaerobic digestion plant was built at a leper colony in India in 1859 (Meynell, 1976). By 1895, biogas from sewage treatment works was used to fuel streetlamps in Exeter, England (McCabe & Eckenfelder, 1957). By the 1930's, developments in the field of microbiology led to the identification of anaerobic bacteria and the conditions that promote methane production. Now, tens of thousands of AD plants are in operation at water treatment plants worldwide.

Landfill gas extraction started in the USA in the early 1970s and spread in Europe, mainly in the United Kingdom and Germany (Braun, 2007). There are currently several thousand landfill gas extraction plants in operation worldwide, representing the biggest source of biogas in many countries.

Anaerobic digestion received renewed attention for agri-industrial applications after the 1970s energy crisis (Ni & Nyns, 1996). When AD was first introduced in the 1970s and 80s, failure rates were very high (Raven & Gregersen, 2007). AD-plant failures were mainly attributed to poor design, inadequate operator training and unfavourable economics (either as a result of unfavourable economies of scale or an unreliable market for biogas). In many parts of the world, these initial experiences have now been overcome with better and more robust reactor designs and with more favourable economic incentives for biogas utilisation.

In developing countries, AD is closely connected with sustainable development initiatives, resource conservation efforts, and regional development strategies (Bi & Haight, 2007; Wang

& Li, 2005). Rural communities in developing countries generally employ small-scale units for the treatment of night soil and to provide gas for cooking and lighting for a single household. Nepal is reported to have some 50,000 digesters and China is estimated to have 14 million small-scale digesters (Wellinger, 2007). Bi & Haight (2007) described a typical household digester in Hainan province (China) to be of concrete construction, about 6m^3 in size and occupying an area of about 14m^2 in the backyard. Digesters are connected with household toilets and the livestock enclosure so that both human and animal manure can flow directly into the digesters. Agricultural straw is also often utilised as feedstock. The digesters are connected to a stove in the house by a plastic pipeline.

Before the introduction of AD, the majority of villagers had relied heavily on the continuous use of firewood, agricultural residues and animal manure in open hearths or simple stoves that were inefficient and polluting. The smoke thus emitted contains damaging pollutants, which may lead to severe illness, including pneumonia, cancer, and lung and heart diseases (Smith, 1993). Combustion of biomass in this way is widespread throughout the developing world and it is estimated to cause more than 1.6 million deaths globally each year (400,000 in Sub-Saharan Africa alone), mostly among women and children (Kamen, 2006). In contrast, biogas is clean and efficient with carbon dioxide, water and digestate as the final by-products of the process. It also conserves forest resources since demand for firewood is lessened when AD is introduced.

2.1 Two models of on-farm anaerobic digestion

Agricultural AD plants are most developed in Germany, Denmark, Austria and Sweden. There are two basic models for the implementation of agriculture-based AD plants in the EU (Holm-Nielsen et al., 2009):

- Centralised plants that co-digest animal manure collected from several farms together with organic residues from industry and townships. These plants are usually large scale, with digester capacities ranging from a few hundred to several thousand cubic meters.
- Farm-scale AD plants co-digesting animal manure and, increasingly, bioenergy crops from one single farm or, sometimes two or three smaller neighbouring farms. Farm-scale plants are usually established at large pig farms or dairy farms.

Centralised AD plants are a unique feature of the Danish bioenergy sector. According to Holm-Nielsen et al. (2009), the Danish AD production cycle represents an integrated system of renewable energy production, resource utilisation, organic waste treatment and nutrient recycling and redistribution. In 2009, there were 21 centralised AD plants and 60 farm-scale plants in Denmark (Holm-Nielsen, 2009). With recent increases in financial incentives provided by the Danish Government, biogas production is expected to triple by 2025 and the number of centralised plants will increase by about 50 (Holm-Nielsen & Al Seadi, 2008; Holm-Nielsen, 2009).

Farm-scale AD plants typically use similar technologies to the centralised plant concept but on a smaller scale. Germany is an undisputed leader in the application of on-farm AD systems with over 4,000 plants currently in operation. The German government also has ambitious plans to expand these numbers even further in order to meet a target of 30% renewable energy production by 2020 (Weiland, 2009). In order to meet this target, the number of AD plants will need to increase to about 10,000 to 12,000. Photovoltaics and wind energy are also widely distributed on farms throughout Germany. It is not uncommon to see an AD plant, a wind turbine and photovoltaics on a single farm (Fig. 1).

Approximately 80% of the biomass used in these plants is manure (mainly slurry), co-digested with 20% organic waste made up of plant residue and agro-industrial waste (da Costa Gomez & Guest, 2004). The biogas is mainly used for combined heat and power (CHP) generation, with the heat generated being used locally for district heating. Biogas is also sometimes up-graded to natural gas quality for use as a vehicle fuel, a practice that is now increasingly common in Sweden (Lantz et al., 2007; Persson et al., 2006).

Fig. 1. "Energy farming in Germany". A single farm is shown here combining an AD plant, wind turbines and photovoltaics on farm buildings. Photo: J. Biala

2.2 Drivers for investment in on-farm anaerobic digestion

Local conditions are particularly important to the decisions of farmers with respect to investing in renewable energy technologies (Ehlers, 2008; Khan, 2005; Raven & Gregersen, 2007). The two most important issues regarding biomass use for energy production in most countries are economic growth and the creation of regional employment. Avoiding carbon emissions, environmental protection and security of energy supply are often big issues on the national and international stage, but the primary driving force for local communities are much more likely to be employment or job creation, contribution to regional economy and income improvement (Domac et al., 2005). The flow-on benefits from these effects are increased social cohesion and stability through the introduction of a new employment and income generating activity.

A range of policy instruments has been used by different countries seeking to develop their renewable energy industries, including renewable energy certificate trading schemes, premium feed-in-tariffs, investment grants, soft loans and generous planning provisions (Thornley & Cooper, 2008). In particular, Germany's generous feed-in-tariffs for renewable energy are typically credited with the massive expansion of on-farm AD plants in that country. Germany introduced the feed-in tariff model in 1991, obliging utilities to buy electricity from producers of renewable energy at a premium price. The feed-in tariff law has been continually revised and expanded. The premium price is technology dependent and is guaranteed for 20 years with a 1% digression rate built in to promote greater efficiency. Investors therefore have confidence in the prospective income from any newly proposed renewable energy project and can develop a more solid business case for obtaining finance.

Whilst the feed-in tariff law has had a marked impact on the diffusion of on-farm AD in Germany, a more complete picture emerges when the underlying political, institutional and socio-economic drivers in the country are considered (Wilkinson, 2011). For example, energy security and climate change mitigation are major geopolitical drivers in Germany. In addition, the impact of the EU's Common Agricultural Policy has been profound in driving both political and grass-roots efforts to develop alternative approaches to farming, including on-farm bioenergy production (Plieninger et al., 2006).

3. Overview of the anaerobic digestion process

The microbiology of the AD process is very complex and involves 4 stages (Fig. 2). The first stage of decomposition in AD is the liquefaction phase or hydrolysis, where long-chain organic compounds (e.g. fats and carbohydrates) are split into simpler organic compounds like amino acids, fatty acids and sugars. The products of hydrolysis are then metabolised in the acidification phase by acidogenic bacteria and broken down into short-chain fatty acids (e.g. acetic, proprionic and butyric acid). Acetate, hydrogen and carbon dioxide are also created and act as initial products for methane formation. During acetogenesis, the organic acids and alcohols are broken down into acetic acid, hydrogen and carbon dioxide. These products act as substrates for methanogenic microorganisms that produce methane in the fourth and final phase called (methanogenesis).

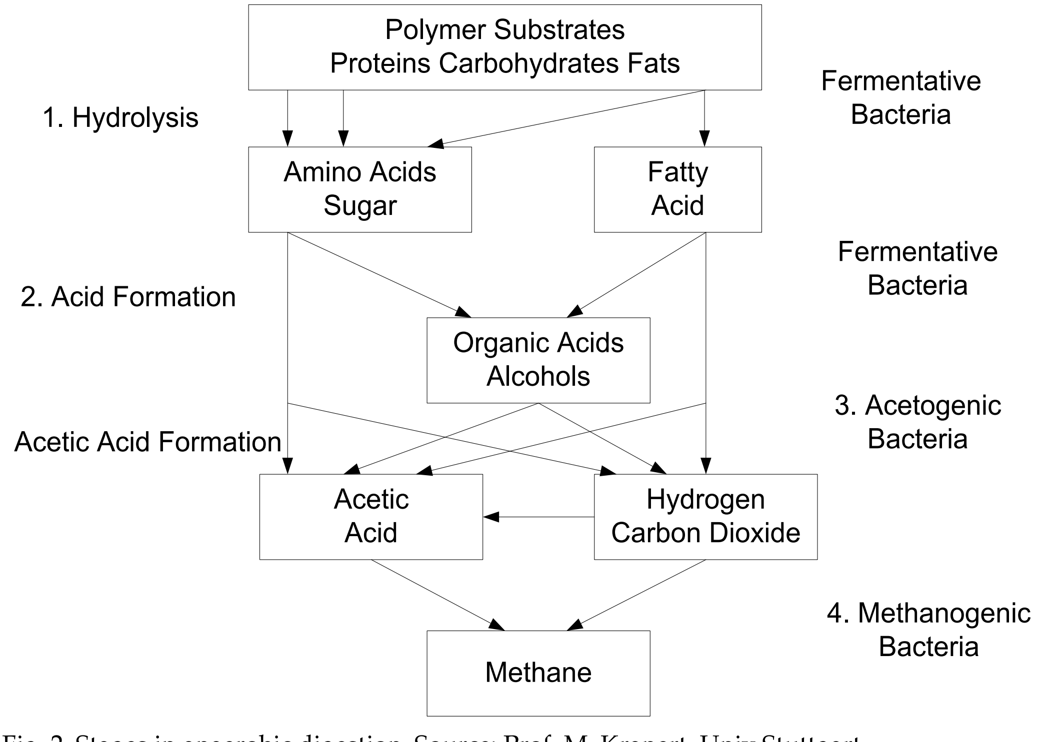

Fig. 2. Stages in anaerobic digestion. Source: Prof. M. Kranert, Univ Stuttgart.

AD systems usually operate either in the mesophilic (35-40ºC) or the thermophilic temperature (50-60ºC) ranges. Operating in the thermophilic temperature range reduces hydraulic retention time (HRT or treatment time) to as low as 3-5 days[1] and more effectively contributes to the sanitisation of the organic waste streams (i.e. improves pathogen and weed-seed destruction). However, greater insulation is necessary to maintain the optimum temperature range, and more energy is consumed in heating thermophilic systems. Larger, centralised systems typically run at thermophylic temperatures. Mesophylic systems need a longer treatment time to achieve good biogas yields but these systems can be more robust than thermophilic systems.

4. Anaerobic digestion systems

AD systems are relatively simple from the process engineering point of view, since fermentation is driven by a "mixed culture" of ubiquitous organisms, and no culture enrichment is generally required (Braun, 2007). Instead, the course of fermentation is controlled by the conditions at start-up: temperature, substrate composition, organic loading rate and hydraulic retention time. Since methane is fairly insoluble in water it separates itself from the aqueous phase and accumulates in the head space of the reactor and is easily collected from there.

A generalised, simplified scheme of the process typical of European systems (Fig. 3) comprises 4 steps:

- substrate delivery, pre-treatment and storage,
- digestion,
- digestate use, and
- energy recovery from biogas.

Usually the effluent leaves the digester by gravity flow and in most cases undergoes further digestion in a second reactor. A tank stores digestate for many months before it is applied directly to farming land. Sometimes the digestate is dewatered prior to undergoing further treatment and disposal (e.g. composting) and the liquid fraction is used as a fertiliser. The head space of the digestate storage tank is typically also connected to the gas collection system. Biogas is collected in both digestion reactors and stored in gas storage tanks or, more frequently in the head space of the second digester, covered with a floating, gas tight membrane. Depending on its final use, biogas can undergo several purification steps. Desulphurisation (to remove corrosive H_2S) is required before the biogas can be combusted in burners or used in combined heat and power (CHP) plants. Desulphurisation can be simply achieved by the controlled addition of air into the digester head space. If biogas is intended for use as a transport fuel or to be fed into the natural gas grid, further upgrading to remove CO_2 is required (Fig. 3).

4.1 System designs

In a batch system, biomass is added to the digester at the start and is sealed for the duration of the process. High-solids systems (total solids content up to 40%) are examples of batch systems. These systems are becoming more widespread for the treatment of municipal

[1] E.g. High-rate anaerobic digestion of waste water. Longer HRTs are typical for semi-solid and solid organic waste streams.

wastes in some parts of Europe (Braun, 2007). In these systems, the solid feedstock is loaded into several reactor cells in sequence. These systems are relatively cheap to construct, require little additional water to operate but the remaining digestate often requires intensive treatment by aerobic composting.

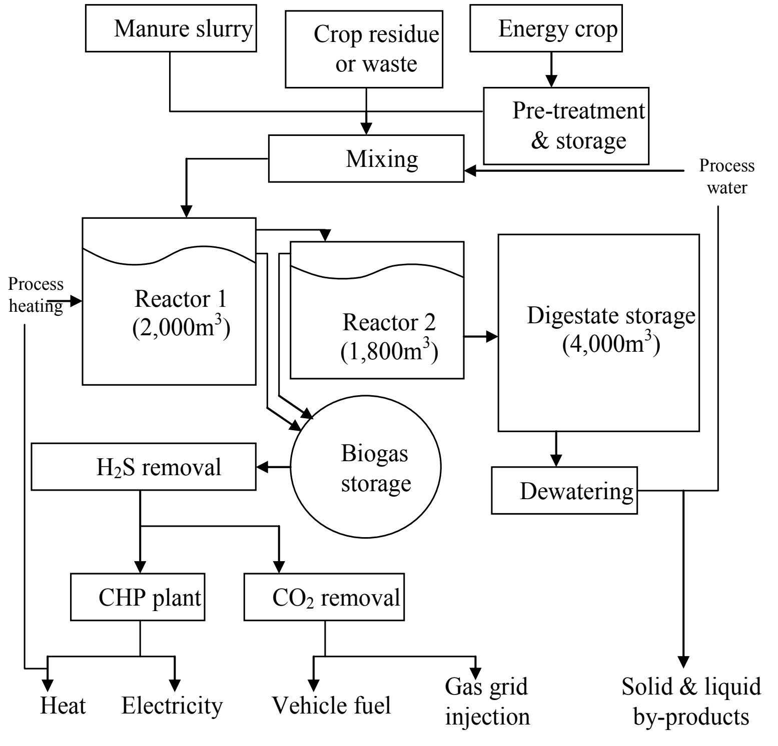

Fig. 3. Typical process-flow diagram for the European 2-stage anaerobic digestion process. CHP – combined heat and power. Source: Wilkinson (2011).

In continuous digestion processes, organic matter is added constantly or in stages to the reactor. Here the end products are constantly or periodically removed, resulting in constant production of biogas. Examples of this form of anaerobic digestion include, covered lagoons, plug-flow digesters, continuous stirred-tank reactors (CSTRs), upflow anaerobic sludge blanket (UASB), expanded granular sludge bed (EGSB) and internal circulation reactors (IC). The most common systems used world-wide for processing manure slurries and agricultural residues are covered lagoons and plug-flow digesters (particularly in North America) and continuous stirred-tank reactors (in Europe and North America). UASB, EGSB and IC reactors are more commonly associated with the anaerobic digestion of wastewater at municipal water treatment plants and will therefore not be discussed in detail here.

Covered lagoon digesters are the cheapest available AD systems. About 19 of the approximately 140 on-farm digesters in the USA are of this type (USEPA, 2009). They can be a viable option at livestock operations in warm climates discharging manure in a flush management system at 0.5-2% solids. The in-ground, earth or lined lagoon is covered with a flexible or floating gas tight cover. Retention time is usually 30-45 days or longer depending on lagoon size. Very large lagoons in hot climates can produce sufficient quantity, quality and consistency of gas to justify the installation of an engine and generator. Otherwise gas production can be less consistent and the low quality gas has to be flared off much of the year.

Plug-flow digesters are also common in the USA where they make up more than half of the on-farm AD plants currently in operation (USEPA, 2009). A plug-flow digester is a long narrow insulated and heated tank made of reinforced concrete, steel or fiberglass with a gas tight cover to capture the biogas. These digesters operate at either mesophilic or thermophilic temperatures. The plug flow digester has no internal agitation and is loaded with thick manure of 11–14% total solids. This type of digester is suited to scrape manure management systems with little bedding and no sand. Retention time is usually 15 to 20 days. Manure in a plug flow digester flows as a plug, advancing towards the outlet whenever new manure is added.

Continuous stirred-tank reactors are most commonly used for on-farm AD systems in Europe (Braun, 2007) and about a quarter of on-farm digesters in the USA are of this type (USEPA, 2009). This type of digester is usually a round insulated tank made from reinforced concrete or steel, and can be installed above or below ground. The contents are maintained at a constant temperature in the mesophilic or thermophilic range by using heating coils or a heat exchanger. Mixing can be accomplished by using a motor driven mixer, a liquid recirculation pump or by using compressed biogas. A gas tight cover (floating or fixed) traps the biogas. The CSTR is best suited to process manure with 3-10% total solids and retention time is usually 10-20 days.

5. Use of digestate

One advantage attributed to farm-based AD systems is the transformation of the manure into digestate, which is reported to have an improved fertilisation effect compared to manure (Börjesson & Berglund, 2003, 2007), potentially reducing the farmer's requirements for commercial fertilisers. The use of digestate instead of commercial fertilisers is also encouraged in Sweden by a tax on the nitrogen in commercial fertilisers (Lantz et al., 2007). However, these incentives are weakened by the limited knowledge and practise of using digestate, as well as the higher handling costs connected with the digestate compared with commercial fertilisers.

In order to control the quality of digested manure, the three main components of the AD cycle must be under effective process control: the feedstock, the digestion process, and the digestate handling/storage (Al Seadi, 2002). The application of digestate as fertiliser must be done according to the fertilisation plan of the farm. Inappropriate handling, storage and application of digestate as fertiliser can cause ammonia emissions, nitrate leaching and overloading of phosphorus. The nitrogen load on farmland is regulated inside the EU by the Nitrates Directive (91/676/EEC nitrate) which aims to protect ground and surface water from nitrate pollution. However, the degree of implementation of the Nitrates Directive in EU member countries varies considerably (Holm-Nielsen et al., 2009).

6. Maximising biogas yields with co-digestion

A key factor in the economic viability of agricultural AD plants is the biogas yield (often expressed as m^3 biogas produced per kg of volatile solids (VS) added). Traditional AD systems based solely on manure slurries can be uneconomic because of poor biogas yields since manure from ruminants is already partly digested in the gut of the animal. Whilst a wide range of substrates can be theoretically digested, biogas yields can vary substantially (Table 1). To put this into perspective, if 1 m^3 of biogas per m^3 of reactor volume is produced per day from digesting manure alone, between 2 to 3 m^3 biogas per m^3 per day can be produced if energy-rich substrates such as crop residues and food wastes are used.

Centralised AD plants receiving agri-industrial and/or municipal wastes as well as farm-based residues also receive an additional gate fee for the wastes they receive. However, where bioenergy crops are grown, economic viability is affected by the cost of growing the crops, any economic incentives provided to grow them and the quality of the final substrate. The cost of supplying energy crops for biogas plants has been increasing in recent years in the EU due to high world food prices rather than competition for land (Weiland, 2008). Data from Germany showed that the cost of supplying maize for silage (minus transport and ensiling) rose 83% between October 2007 and October 2008 (Weiland, 2008).

Although co-digestion with energy crops is not a new concept, it was first considered not to be economically feasible (Braun, 2007). Instead, crops, plants, plant by-products and waste materials were added occasionally just to stabilise anaerobic digesters. However, with steadily increasing oil prices and the improved legal and economic incentives emerging in the 1990s, energy crop R&D was stimulated, particularly in Germany and Austria. Now, 98% of on-farm digesters in Germany utilise energy crops as a substrate (Weiland 2009).

Organic material	Biogas yield (m^3/kg VS)	Min HRT* (d)
Animal fat	1.00	33
Flotation sludge	0.69	12
Stomach- and gut contents	0.68	62
Blood	0.65	34
Food leftovers	0.47-1.1	33
Rumen contents	0.35	62
Pig manure	0.3-0.5	20
Cattle manure	0.15-0.35	20
Chicken manure	0.35-0.6	30
Primary industrial sewage sludge	0.30	20
Market waste	0.90	30
Waste edible oil	1.104	30
Potato waste (chips residues)	0.692	45
Potato waste (peelings)	0.898	40
Potato starch processing	0.35-0.45	25
Brewery waste	0.3-0.4	14
Vegetable and fruit processing	0.3-0.6	14

*HRT – hydraulic retention time (ie duration of processing before stabilization)

Table 1. Biogas yields from various organic materials conducted in batch tests. Source: Braun (2007).

A wide variety of energy crops can be grown for anaerobic digestion, but maize is by far the most important and it also has a higher potential biogas yield per ha cultivated than most other crops (Hopfner-Sixt & Amon, 2007; Weiland, 2006; Table 2). Since the key factor to be optimised in biogas production is the methane yield per ha, specific harvest and processing technologies and new genotypes will increasingly be used when crops are required as a renewable energy source.

In order to maintain a year-round supply of substrate to the digester, the harvested energy crop must be preserved by ensiling. Optimal ensiling results in rapid lactic acid (5–10 %) and acetic acid fermentation (2–4%), causing a decrease of the pH to 4–4.5 within several days (Braun et al., 2008). Silage clamps or bags are typically used. Improper preparation and storage of silage is critical to successful utilisation in AD plants. For example, Baserga & Egger (1997; cited in Prochnow et al., 2009) demonstrated a remarkable reduction in biogas yields due to aerobic deterioration of grass silage. Immediately after opening of a silage bale the biogas yield was 500 L/kg DM, after five days 370 L and after 30 days only 250 L. Similarly, biogas yields from grass silage cut in summer in southeast Germany produced 216 L/kg DM for a well preserved silage but 155 L for spoiled silage (Riehl et al., 2007; cited in Prochnow et al., 2009).

Special care must also be taken in case of substrate changes. Changing composition, fluid dynamics and bio-degradability of the substrate components can severely impede digestion efficiency resulting in digester failures (Braun et al., 2008). Large scale commercial energy crop digestion plants mainly use solid substrate feeding hoppers or containers for dosing the digester continuously via auger tubes or piston pumps. Commonly energy crops are fed together with manure or other liquid substrates, in order to keep fermentation conditions homogenous.

Crop	Biogas yield (m^3/t VS)	Crop	Biogas yield (m^3/t VS)
Maize (whole crop)	205 – 450	Barley	353 – 658
Wheat (grain)	384 – 426	Triticale	337 – 555
Oats (grain)	250 – 295	Sorghum	295 – 372
Rye (grain)	283 – 492		
Grass	298 – 467	Alfalfa	340 – 500
Clover grass	290 – 390	Sudan grass	213 – 303
Red clover	300 – 350	Reed Canary Grass	340 – 430
Clover	345 – 350	Ryegrass	390 – 410
Hemp	355 – 409	Nettle	120 – 420
Flax	212	Miscanthus	179 – 218
Sunflower	154 – 400	Rhubarb	320 – 490
Oilseed rape	240 – 340	Turnip	314
Jerusalem artichoke	300 – 370	Kale	240 – 334
Peas	390		
Potatoes	276 – 400	Chaff	270 – 316
Sugar beet	236 – 381	Straw	242 – 324
Fodder beet	420 – 500	Leaves	417 – 453

Table 2. Typical methane yields from digestion of various plants and plant materials as reported in literature (Data compilation after Braun, 2007)

The total solids content of feedstock in these systems is usually <10% and mechanical stirrers are used for mixing. The typical two-digester, stirred tank design described above is used in most of these digestion plants. Anaerobic digestion of energy crops requires hydraulic retention times from several weeks to months. Complete biomass degradation (80-90% of VS) with high gas yields is essential to maintain the economic viability and environmental performance of the digestion process.

7. Improving energy efficiency

Combustion in burners for heating purposes is the simplest application for the energy content of biogas, and this can be achieved with comparably high efficiency. Alternatively, biogas is converted into electrical energy by the use of an engine and generator. Combined heat and power (CHP) plants are widely used in AD plants though waste heat is generally under-utilised. It is widely agreed that increased use of waste heat in CHP plants is critical for the long-term economic and environmental performance of AD plants. This is especially the case where the costs of energy crops as feedstock have risen concomitantly with the rapid diffusion of AD plants, for example in Germany (Weiland, 2009).

The use of biogas in CHP simultaneously transfers the chemical energy of methane into electrical power (about 1/3rd) and heat (about 2/3^{rds}). CHPs often result in low overall energy efficiencies because the degree of heat use in many cases is quite small. Of a survey of 41 Austrian digestion plants, CHP energy efficiency ranged from 30.5 to 70.7% (Braun et al., 2008).

Nevertheless, there are examples of the effective use of waste heat in Scandinavian countries where district heating grids are more commonplace (Holm-Nielsen et al., 2009). And in Germany, municipal authorities have developed district heating CHP systems to provide heat and power to businesses and residents in many cities for >100 years (Kerr, 2009).

There is a wide range of CHP technologies commercially available, such as diesel engines converted to run on dual-fuel, gas turbines and Stirling engines (Lantz et al., 2007). These applications are available in size from approximately 10kW$_{el}$ to several MW$_{el}$. Small-scale CHP may prove to be suitable at small, farm-based AD plants although scale effects and the problems concerning the utilisation of the heat discussed above make large-scale applications more economical under current conditions (Lantz et al., 2007).

8. Upgrading of biogas for use in vehicle fuels or natural gas grids

In the EU countries where AD is well-established, upgrading of biogas is increasingly being considered so that it can be injected into the natural gas grid or used as a vehicle fuel. Before biogas is suitable for these applications, it must be upgraded to natural gas quality by the removal of its CO_2 content and other contaminants (e.g. H_2S, NH_3, siloxanes and particulates). Commercially available technologies available to remove CO_2 include pressurized water absorption and pressure swing adsorption.

In response to CO_2 emission reduction targets, the EU biofuels directive set a target of replacing 5.75% of transport fuels with biofuels by 2010. Up to date we have seen a rapid increase in bioethanol and biodiesel production since commercial conversion technologies, infrastructure for distribution, and vehicle technologies, currently favour these types of biofuels (Börjesson & Mattesson, 2007). Their competitiveness has also increased with an increase in the price of crude oil. The production costs of using upgraded biogas as a vehicle fuel in the EU are in the same ball-park as wheat-based ethanol and biodiesel from vegetable oils (Börjesson & Mattesson, 2007). But owing to the increased costs associated with adapting vehicles to run on biogas (+10% to new car prices), its price needs to be 20–30% lower than the price of other vehicle fuels.

However, the use of biogas in this manner has several advantages over bioethanol and biodiesel:

- The net annual energy yield per hectare from the AD of energy crops is potentially about twice that of bioethanol from wheat and biodiesel from rapeseed.
- AD could be integrated with bioethanol and biodiesel production to improve their overall resource efficiency by using their by-products to produce biogas.
- Net greenhouse gas (GHG) savings from the use of biogas as fuel could approach 140-180% due to the dual benefit of avoided emissions from manure storage and the replacement of fossil fuels. In comparison, the likely savings in GHG emissions from biodiesel and bioethanol production and use are much lower.[2]

A prominent example of upgrading biogas and using it for vehicle fuel is Sweden, where the market for such biogas utilisation has been growing rapidly in the last decade. Today there are 15,000 vehicles driving on upgraded biogas in Sweden, and the forecast is for 70,000 vehicles, running on biogas supplied from 500 filling stations by 2012 (Persson et al., 2006). In Sweden, the production of vehicle fuel from biogas has increased from 3TJ in 1996 to almost 500 TJ in 2004 or 10% of the current total biogas production. Yet this corresponds to only 0.2% of Sweden's total use of petrol and diesel.

Germany and Austria have also recently set goals of converting 20% biogas into compressed natural gas by 2020 for more efficient use in CHP systems, gas network injection or vehicle fuel use (Persson, 2007). Weiland (2009) predicts that about 1,000 biogas upgrading plants will be needed to meet the government's objective with a projected investment of €10 billion required. To achieve these targets, the German government has developed a comprehensive program of financial incentives. Germany also currently has the largest biogas upgrading plant in the world located at Güstrow with a capacity of 46 million m^3.

9. Conclusion

The threats of climate change, population growth and resource constraints are forcing governments to develop increasingly stronger policy measures to stimulate the development of renewable energy technologies. Bioenergy offers particular promise since it has the potential to deliver multiple benefits such as: improved energy security, reduced CO_2 emissions, increased economic growth and rural development opportunities. Anaerobic digestion is one of the most promising renewable energy technologies since it can be applied in multiple settings such as wastewater and municipal waste treatment as well as in agriculture and other industrial facilities.

Increasing the efficiency of converting biomass to utilisable energy (ie heat and electricity) is critical for the long-term environmental and financial sustainability of AD plants. Even with

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[2] Under Scandinavian conditions where the heat and electricity used in bioethanol and biodiesel plants are generated from renewable sources, the GHG savings could range from 60 to 90%. Where these plants use fossil fuels for heating and electricity, the GHG benefits will be much lower.

generous incentives such as those provided by many EU governments, increasing construction costs and the rising cost of energy crops can put the financial viability of AD plants at risk. Unless improvements in efficiency are found and implemented, these pressures could lead to unsustainable rises in the cost of the incentive schemes that underpin the development of renewable energy technologies.

9.1 Future work

Landscapes that are dominated by arable agriculture have always been subject to change, but increasing concerns over energy security and climate change could precipitate major land-use changes on large areas of land over relatively short time-scales. The impact of a rapidly expanding bioenergy industry in many countries is already under scrutiny due to the emergence of a number of unintended consequences. The unintended consequences include competition for food and land, indirect land use change, and landscape scale impacts on water, biodiversity and social values. Consequently, sustainability assessment systems are now beginning to be developed, and institutional systems are being used to set sustainability targets rather than just to stimulate industry expansion (O'Connell et al., 2009).

Systems need to be developed to monitor and deal with sustainability issues at the local level. In particular, there is a need to explore the sustainability of different pathways for industry development and growth. An important part of this process is to develop the tools to assess the inevitable trade-offs that will result between the different components of sustainability.

In addition to the broader consideration of sustainability, R&D needs that are specific to on-farm AD systems include:

- Developing cost-effective AD systems that are purpose designed for different applications (both large-scale and small scale). The capital cost of many on-farm AD systems has been increasing in recent years and could be over-engineered for many applications.
- Developing new higher-yielding energy crops that use less water, pesticides and fertiliser inputs. These crops should not directly compete with food crops and could be grown on under-utilised farming land.
- Conducting studies to increase the conversion efficiency of energy crops to biogas.
- Improving CHP technologies and distribution systems for utilising waste heat for different heating and cooling applications.

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11

New Municipal Solid Waste Processing Technology Reduces Volume and Provides Beneficial Reuse Applications for Soil Improvement and Dust Control

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1. Introduction

The disposal of municipal solid waste (MSW) is an ongoing problem in the United States, and even US Army installations, as a microcosm representing small to medium sized cities across the country, are not immune. The Army generated over 1.2 million metric tons of solid waste in the United States in Fiscal Year 2003 (Solid Waste Annual Reporting, 2003), with a majority shipped to off-post landfills at considerable expense. The US Army Environmental Command has ranked solid waste management as the number one pollution prevention challenge for the Army. Like most cities and towns in the United States and other developed countries, the military is faced with decreasing landfill space, increasing costs of disposal, and mounting environmental pressures for remediation of leaking landfills. The Army operates 16 active landfills that have less than 10 years of useful life, and current landfill costs exceed $140 million annually. These costs are expected to increase dramatically over the next several years with the added pressures of mandated military environmental stewardship and remediation liability for older landfills that have started to leak.

Faced with Executive Order 13101, which dictates Government strategies for waste prevention, recycling, and Federal acquisition policy, and, memorandum DUSD (ES), May 13, 1998, specifying a landfill diversion rate of 40% by 2005, the Government must immediately develop and exploit technologies capable of satisfying these requirements. MSW represents approximately 20% of the total solid waste streams generated by military installations [Waste and Recycling, Jul. 2002]. A major reduction in the amount of this material landfilled contributes significantly, agency-wide, to reaching the goal of Executive Order 13101.

One possible method to relieve this waste problem is to reduce the volume of the municipal solid waste or utilize waste in methods other than landfilling. Collection and composting of organic food, processing, and landscape wastes, as well as paper, glass and metal recycling have made significant contributions to reducing waste volume, but landfill utilization requirements still exceed landfill capacity. Processes and equipment to facilitate the rapid separation, volume reduction, and conversion into reusable products have been developed and tested in limited capacities. This chapter describes one such process developed by

Bouldin and Lawson, Incorporated, which produces a cellulosic by-product, trademarked as Fluff®, which has been shown to be suitable for a number of uses based on laboratory test results that indicate it is relatively benign from an environmental aspect despite the municipal input stream from which it is derived. A full spectrum of research will be presented to highlight (1) the chemical and agronomic nature of this material called Fluff, (2) its mineralization characteristics when used as a soil additive, (3) the effects of land application on vegetative plant growth, development, biomass production, and chemical composition of plant tissues, (4) the effects of land application on numerous soil chemical and physical properties, and (5) results from a proof of concept study showing the applicability of this material for use as a dust suppressant on unpaved roads.

2. Fluff: a unique municipal solid waste processing by-product

A solid waste processing technology that facilitates the rapid separation, volume reduction, and conversion of municipal waste into a sterile organic pulp has been developed and deployed at several locations. This process separates the organic fraction of municipal garbage from the recyclable materials and then sterilizes it, producing a pulp-like material called Fluff® (Bouldin & Lawson, Inc., 2000). Raw municipal refuse including paper, glass, metals, plastics, wood, food wastes, vegetative wastes, and other inert materials are introduced into a low-speed, high-torque shredder where the materials are reduced into approximately 2-5 cm square pieces. Batteries, carpet, and any other items that might cause equipment or personnel harm are typically removed by hand from the input stream. The shard pieces produced by the high-torque shredder are delivered to a conveyor system that utilizes magnetic rollers to separate out the ferrous metals. The balance of the waste is then further reduced using a series of smaller shredders and grinders before being conveyed into a hydrolyzer, a jacketed containment vessel using high temperature steam in a proprietary process [US Patent No. 6,017, 475] to break molecular bonds and destroy pathogens (Bouldin & Lawson, Inc., 2000). The resultant hydrolysis product is transferred to an expeller unit (auger) that operates as a "hard" press, serving as a ram to shuttle the moist cellulosic material along an internally tapered tunnel. Water is then removed from the aggregate cellulose in a rotary dryer. The coarse and fine cellulosic mixes are separated from one another through the use of screens and compressed air classification; the lighter, coarser material is deposited in a collection bin while the smaller fractions are tumbled through a rotary drum to remove the fines of aluminum, glass, and plastic, which are gravity-fed into a "particulates" collection bin. The separated fine cellulose material emerges as a sanitized, sand-like granular fluff that is useful as a soil amendment because of its organic base and relatively high nitrogen content. If not utilized as a soil amendment, the Fluff byproduct can still be landfilled at a 30-75% reduction in volume, depending upon input materials (BouldinCorp, unpublished data, 2001). The coarse, peat moss-like material can be extruded into plastic-like composite planks.

This technology is currently in use in Warren County, TN, where a 95% recycling rate has been achieved for the county's municipal solid waste, with the bulk of the organic byproduct composted for use as a topsoil replacement in the horticultural industry (Croxton et al., 2004). Several processing systems have also been deployed in the island countries of Aruba, St. Croix, and St. Thomas, where land fill space is at a premium and alternative Fluff uses include pelleted fuel production and beach erosion prevention and restoration.

Fig. 1. Picture of Fluff material.

3. Fluff analysis

The Fluff byproduct has been analyzed for nutrient components important to agriculture and found to have significant nutrient concentrations that could serve as an organic fertilizer source (Table 1) (Busby et al., 2006; Busby, 2003). Fluff has also been intensively analyzed for levels of 184 regulated compounds, including 11 heavy metals, 113 semi-volatile and 60 volatile organic compounds to determine any potential regulatory limitations. Analyses of toxicity characteristic leaching procedure (TCLP) volatiles, TCLP semi-volatiles, TCLP heavy metals, TOX (total organic halogen content), and low resolution dioxin content were performed by PDC Laboratories (Peoria, IL), a United States Environmental Protection Agency (EPA) certified laboratory for Tennessee, using EPA methods SW846-8260, 8270, 1311, 9076, and 8280, respectively (USEPA 1998). Only 9 heavy metals, 3 semi-volatile and 3 volatile organic compounds were detected. The detected organic compounds [acetone, methylene chloride, toluene, di(2-ethylhexyl)phthalate, di-n-butyl phthalate, and di-n-octyl phthalate] are regulated in either the Clean Water Act or the Clean Air Act due to risks associated with workplace exposure and concentrated industrial effluent. However, due to their volatile chemical nature and rapid turnover in the environment, they pose very little risk at concentrations found in the Fluff, especially when incorporated into the topsoil, and therefore are not regulated for this purpose.

Limits have been established for land application of heavy metals in biosolids and these existing standards were used to assess metal loading of Fluff in the absence of a similar compost standard (40 C.F.R. Part 503, 1999). A comparison of Fluff heavy metal concentrations and EPA biosolids limits for maximum metal concentrations, maximum annual soil metal loading, and maximum cumulative soil metal loading are presented in Table 2. In comparing metal concentrations in Fluff to the biosolids ceiling limits, it was found that all Fluff metal concentrations were at least an order of magnitude below their respective ceiling limits. The Fluff metal concentrations were used to calculate maximum annual and cumulative application rates, where lead (Pb) was found to be the contaminant of primary concern. Annually, this limit would be reached with an application rate of 229

Mg ha^{-1}. The maximum cumulative Fluff application rate was found to be 4587 Mg ha^{-1}, or 20 repeated applications at the maximum annual limit. However, other factors would most likely preclude achieving these rates due to material and land availability, transportation, and effective soil incorporation constraints. Agriculturally significant properties of the Fluff are presented in Table 1. Fluff has a near-neutral pH a C:N ratio around 30, and research indicates it decomposes slowly (Busby et al., 2007).

pH	6.5
C:N	32
C (%)	39.8
N (%)	1.26
P (mg kg^{-1})	1900
K (mg kg^{-1})	2170
Ca (mg kg^{-1})	13600
Mg (mg kg^{-1})	1400
Fe (mg kg^{-1})	2460
Mn (mg kg^{-1})	130
Zn (mg kg^{-1})	234
B (mg kg^{-1})	35
Cu (mg kg^{-1})	47.7
Co (mg kg^{-1})	2.0
Na (mg kg^{-1})	5169

Table 1. Fluff properties significant to agriculture.

3.1 Fluff C mineralization analysis

A key component of this new municipal solid waste processing technology is that Fluff can be utilized as a soil amendment to improve soil physical and chemical condition. Since most contaminants and pathogens are removed through the processing technology, Fluff could bypass the composting process and eliminate the most negative aspects of large-scale composting: the time, facilities infrastructure, and resulting management costs as well as the associated problems with leachate, odors, pests, and pathogen exposure (Busby et al., 2003). However, non-composted materials are generally not used because undecomposed organic matter is often attributed to phytotoxic effects and nutrient immobilization when applied to soil (Edwards, 1997; Zucconi et al., 1981a; Chanyasak et al., 1983a,b; Wong, 1985; Bengston and Cornette, 1973; Terman et al., 1973).

When applying organic materials such as municipal waste compost to soil, care must be taken not to adversely affect the establishment and growth of vegetation. Undecomposed compost that is high in NH_4, organic acids, and other compounds can be phytotoxic (Zucconi et al., 1981b; Chanyasak et al., 1983a,b; Wong, 1985). Fortunately, these chemicals most often occur for short durations and do not induce lasting toxic effects in the environment (Zucconi et al., 1981a). Longer term effects can occur from unstabilized organic material with a high C:N ratio, as microbial decomposition can immobilize significant amounts of N, making it unavailable for plant utilization and leading to deficiency problems (Bengston and Cornette, 1973; Terman et al., 1973). Composting organic matter will alleviate these potential problems, but only if the substrate is allowed to compost to maturity.

Metal	Fluff (mg kg^{-1})	Biosolids Ceiling Limits (mg kg^{-1})[2]	Biosolids Annual. Loading Rate Limits (kg ha^{-1} yr^{-1})[3]	Calculated Maximum Annual Fluff Application Rate (Mg ha^{-1} yr^{-1})	Biosolids Cumulative Loading Limits (kg ha^{-1})[4]	Calculated Maximum Cumulative Fluff Application (Mg ha^{-1})
As	<RL[1]	75	2	-	41	-
Ba	46.6	-	-	-	-	-
Cd	1.9	85	1.9	1000	39	20526
Cr	39.8	-	-	-	-	-
Cu	47.7	4300	75	1572	1500	31447
Hg	0.547	57	0.85	1554	17	31079
Ni	9.12	420	21	2303	420	46053
Pb	65.4	840	15	229	300	4587
Se	9.67	100	5	517	100	10341
Zn	234	7500	140	598	2800	11966

[1] Reaction Limit; [2]from 40 CFR Part 503.13, Table 1; [3]from 40 CFR Part 503.13 Table 4; [4]from 40 CFR Part 503.13 Table 2

Table 2. Comparison of fluff heavy metal concentrations with USEPA limits for biosolids application.

To date, there is no agreed upon test to determine whether or not compost is mature, or even what the definition of mature should be. Numerous tests and measurements have been developed to provide insight into changes that occur in decomposing organic matter, how these changes relate to maturity, and how these changes might affect plant growth (Jimenez and Garcia, 1989; Bernal et al., 1998; Cooperband et al., 2003). The general consensus is that a series of tests, encompassing several different aspects of decomposition, is the most reliable method of determining compost maturity. Two validated, simple, and widely used tests to predict the maturity of composting organic matter are the C evolution and N mineralization tests. When combined, these tests can provide good indications of when a decomposing material may be phytotoxic, immobilizing nutrients, or mature by combining measurements of microbial respiration with NO_3 and NH_4 concentrations.

Utilization of non-composted waste material such as Fluff provides significant benefits over handling costs associated with composting. However, because this municipal waste product is unstabilized, there were concerns regarding the effect microbial decomposition of this material has on nutrient availability when used as a soil amendment. Therefore, studies were conducted to determine the rate of decomposition and N cycling of Fluff at increasing rates in two distinct sandy soils and comparing them to mature commercial municipal waste compost (Busby et al., 2007).

In the Busby et al. (2007) study, a 90 d incubation was performed to measure C and N mineralization of composted and un-composted municipal wastes at varying rates. Total C and N of both organic additives were analyzed and indicated that Fluff had a much higher C content but similar nitrogen concentration compared to the compost (Table 3).

	Fluff[1]	MWC[2]
pH	6.5	7.1
C:N	32	12.9
C (%)	37.1	16.5
N (%)	1.26	1.25
P (mg kg^{-1})	1900	3880
K (mg kg^{-1})	2170	7070
As (mg kg^{-1})	<RL	12.2
Cd (mg kg^{-1})	1.9	7.76
Cu (mg kg^{-1})	47.7	811.8
Ni (mg kg^{-1})	9.12	67.2
Pb (mg kg^{-1})	65.4	447.5
Se (mg kg^{-1})	9.67	1.25
Zn (mg kg^{-1})	234	1692

[1]Fluff = Un-composted municipal waste; [2]MWC= Municipal waste compost

Table 3. Chemical properties of organic additives.

Two soils were collected from study sites at Fort Benning, GA: a Troup loamy fine sand (Loamy, kaolinitic, thermic Grossarenic Kandiudults) designated "Dove Field" and a highly disturbed Orangeburg loamy sand (Fine-loamy, kaolinitic, thermic Typic Kandiudults) designated "Borrow Pit" to use in the incubation (Soil Series Classification Database, 2008). Methods of incubation followed techniques described by Torbert et al. (1998). Treatments consisted of 25 g dry weight of sieved soil samples placed in small plastic cups. Composted (MWC) or un-composted municipal waste (Fluff) was mixed into soils at desired application rates, and deionized water was added to moisten soils to 85% of field capacity. Cups were then placed in 1.06 l jars which were fitted with CO_2 traps and incubated in the dark at 25°C and 70% relative humidity for 90 d. Soil samples were extracted every 30 d and analyzed for NH_4 and NO_3. Percent of additive total organic carbon (TOC) mineralized was also determined by subtracting the evolved C for each control soil from each respective experimental unit and dividing the net additive C evolved by the TOC added by each respective additive-rate treatment.

Carbon mineralization of the Fluff was much higher than in the mature compost (Table 4). The higher rates of Fluff application still had significantly higher C evolution rates than the compost even after 90 d of incubation, indicating that this material was still not completely stabilized at a level similar to that of the compost. Further, the percent added TOC mineralized indicates that the compost was much more stable, although the percent Fluff TOC mineralized in the Dove Field soil did stabilize after 60 d (Table 5). Soil type heavily influenced C evolution of the decomposing Fluff, as the soil with higher initial C and N concentrations had significantly higher rates of C evolution across application rates. This difference was most likely due to the differences in available soil N, as the N was extremely limited in the Borrow Pit soil and slightly less so in the Dove Field soil, which could have significantly reduced microbial activity. This is further demonstrated by the inverse relationship between Fluff application rate and percent TOC mineralized in the Dove Field soil, as the N limitation would have increased with increasing Fluff application.

	mg CO2-C kg-1 soil-1 d-1							
	Dove Field Soil							
	17.9 Mg ha-1		35.8 Mg ha-1		71.6 Mg ha-1		143 Mg ha-1	
Day	Fluff*	MWC**	Fluff	MWC	Fluff	MWC	Fluff	MWC
30	24.74	3.50	34.02	4.20	56.12	6.65	80.85	9.81
60	9.94	0.53	17.76	1.93	24.74	1.93	35.10	2.98
90	5.98	1.58	9.64	2.10	13.97	2.45	17.63	4.90
	Borrow Pit Soil							
	17.9 Mg ha-1		35.8 Mg ha-1		71.6 Mg ha-1		143 Mg ha-1	
Day	Fluff	MWC	Fluff	MWC	Fluff	MWC	Fluff	MWC
30	8.72	2.80	21.54	2.80	36.30	4.90	76.08	7.71
60	4.22	1.05	6.61	2.10	13.57	0.18	26.39	2.63
90	0.11	0.00	3.07	0.00	4.65	2.28	16.85	3.68

* Fluff = Un-composted municipal waste
** MWC = Municipal waste compost

Table 4. Comparison of carbon evolution rates between soils, additives, rates, and incubation duration.

Total inorganic N and NO_3 levels were considerably higher in the compost treatments than in the Fluff treatments, indicating that decomposition of the Fluff resulted in significant N immobilization (Table 6). No changes in inorganic N concentration were observed in the Borrow Pit Fluff treatments through 90 d, but the Dove Field Fluff treatments did increase slightly over time, with an inverse relationship between rate and inorganic N concentration after 90 d of incubation. Ammonia levels did not differ at the same magnitude. Ammonia concentrations in the compost treatments remained very low and relatively constant across rates and soils but decreased slightly over time. Ammonia concentrations in the Dove Field

	% C mineralized of additive TOC							
	Dove Field							
	17.9 Mg ha-1		35.8 Mg ha-1		71.6 Mg ha-1		143 Mg ha-1	
Day	Fluff*	MWC**	Fluff	MWC	Fluff	MWC	Fluff	MWC
30	15.45	1.06	11.68	1.14	9.58	1.49	7.43	1.41
60	26.71	1.71	19.27	2.55	15.06	2.19	11.31	2.00
90	25.85	3.30	19.86	3.06	17.83	2.65	12.36	2.64
	Borrow Pit							
	17.9 Mg ha-1		35.8 Mg ha-1		71.6 Mg ha-1		143 Mg ha-1	
Day	Fluff	MWC	Fluff	MWC	Fluff	MWC	Fluff	MWC
30	4.78	0.94	6.46	0.54	6.02	1.13	6.47	1.04
60	9.70	2.18	10.32	2.07	9.22	1.16	9.16	1.51
90	8.91	2.18	11.32	2.07	10.29	2.07	10.34	2.23

* Fluff = Un-composted municipal waste; ** MWC = Municipal waste compost

Table 5. Comparison of percent carbon mineralization of additive total organic carbon (TOC) between soils, additives, rates and incubation duration.

Fluff treatments peaked at day 60 and decreased to their initial levels by day 90, indicating that net ammonification had occurred during the incubation but net nitrification had begun by the end of the 90 d. Even at the peak, however, NH_4 levels still remained at low concentrations (<11 mg kg^{-1}). The low concentrations of NH_4 indicate that potential toxicity from NH_4 buildup would not be a problem in these soils even at rates of 143 Mg ha^{-1}. In the Borrow Pit soil, neither net ammonification nor nitrification was ever indicated throughout the incubation as both NH_4 and NO_3 concentrations stayed consistently low. This consistency indicates a severe N deficiency in this soil and was probably responsible for the slower decomposition of the Fluff.

	Total Inorganic N Concentration (mg kg^{-1})							
	Dove Field Soil							
	17.9 Mg ha^{-1}		35.8 Mg ha^{-1}		71.6 Mg ha^{-1}		143 Mg ha^{-1}	
Day	Fluff*	MWC**	Fluff	MWC	Fluff	MWC	Fluff	MWC
30	2.60	38.54	3.71	59.65	4.17	85.22	3.778	117.68
60	14.89	51.80	12.16	67.34	10.13	108.06	6.586	139.94
90	22.93	56.12	15.35	68.78	7.47	101.57	5.218	132.688
	Borrow Pit Soil							
	17.9 Mg ha^{-1}		35.8 Mg ha^{-1}		71.6 Mg ha^{-1}		143 Mg ha^{-1}	
Day	Fluff	MWC	Fluff	MWC	Fluff	MWC	Fluff	MWC
30	0.00	18.54	0.00	29.11	0.00	55.93	0.50	112.92
60	0.40	17.30	0.93	30.61	1.08	64.55	1.01	123.58
90	0.00	14.64	0.30	31.68	0.11	61.86	0.63	123.79

* Fluff = Un-composted municipal waste; ** MWC = Municipal waste compost

Table 6. Differences in total inorganic nitrogen concentration between soils, additives, rates, and incubation duration.

Because both soils were relatively infertile and both C and N mineralization of the Fluff were closely tied to the fertility status of the soils, it is likely that Fluff decomposition will occur at a faster rate in more fertile soils. When used in infertile soils, N immobilization will occur for an extended period due to incorporation into microbial biomass, with potential negative consequences for vegetation initially, but fertilization with a readily available N source may alleviate the period of this immobilization. On the other hand, slower degradation of the material may provide the best long term benefit as leaching losses would be minimized and N inputs would more closely resemble natural soils, as was found with yard waste compost that led to net immobilization initially (Claassen and Carey, 2004). For vegetation that requires significant N inputs, the mature compost would work well as it provided a steady and significant amount of N throughout the 90 d. In settings where available N could be detrimental, such as native plant restorations or in other instances where weed pressure is undesirable and detrimental, Fluff application could be a simple way to decrease available N in the short term, but would most likely provide a slowly available source over the longer term. Restoration of late-seral plant communities has previously been achieved through high C:N organic soil amendments such as sucrose and sawdust that limit available N (McLendon and Redente, 1992; Morgan, 1994; Paschke et al., 2000). Additionally, any increase in the organic C content of soil can provide significant benefits, especially in degraded soils where vegetative cover is minimal. Soil organic matter reduces compactibility (Zhang et al., 1997), increases water holding capacity (Hudson, 1994), increases particle aggregation (McDowell and Sharpley, 2003), and reduces erodibility (Gilley and Risse, 2000; Barthes et al., 1999).

The comparison between these data and other studies using raw household waste (Bernal et al., 1998) indicates that the MWC used here had a much lower rate of C mineralization relative to the unprocessed waste in the previous study, with the only major difference between the organic materials being the processing technology used to produce MWC. Because the MWC had such a low rate of C mineralization relative to the raw waste, the processing must have a significant effect on the material's degradation rate. If the carbonaceous material resulting from this process increases the residence time of added C in soil, this could be a significant benefit for increasing organic matter in soils. The increase in soil C and decrease in soil N from the un-composted Fluff indicates that it would be best suited for highly degraded soils where establishment of native perennial communities adapted to N limitation is desired.

4. Fluff uses

This waste processing technology is currently in use in Warren County, TN, where a 95% recycling rate has been achieved for the county's municipal waste, with the bulk of the organic byproduct composted for use as topsoil replacement in the horticultural industry (Croxton et al., 2004). While the resulting Fluff material has been used successfully after composting in the horticulture industry, Fluff may also be an effective soil amendment before composting to improve soil physical and chemical properties, thereby enhancing land rehabilitation efforts. The Fluff is unique in both origin and physical attributes when compared to other soil amendments, and land application studies have recently been conducted by the US Army Corps of Engineers to improve Army training ground rehabilitation, based on results of the incubation studies described above. The United States Army generated over 1.2 million metric tons of solid waste in the United States in Fiscal Year 2003 but has a limited number of landfills, increasing costs to ship garbage off post (Solid Waste Annual Reporting, 2004). However, with almost 5 million hectares of land in the United States, including 73 installations with greater than 4,000 hectares each, the Army has enough acreage to support large-scale land utilization of organic waste byproducts (DoD, 2001). Large blocks of this land are in need of rehabilitation due to historic and contemporary Army training activities, but often lack sufficient topsoil, organic matter, and nutrients required for successful rehabilitation. By diverting organic matter from landfills to degraded training lands, the Army could incorporate reuse of municipal waste into land management, decrease waste disposal costs, and improve land rehabilitation efforts on Army training and testing ranges.

Due to the expenses involved with overcoming these land rehabilitation limitations, a cheap alternative material is needed. An effort to utilize organic waste byproducts by the Army could be greatly enhanced if the need for large scale composting facilities for municipal waste could be eliminated. The use of a highly processed organic pulp such as Fluff could divert organic matter from landfills to degraded training lands. On marginal lands such as degraded training areas, organic amendments such as Fluff can be beneficial when used to enhance vegetation establishment. The increased soil organic matter should increase the soil water holding capacity and pH, lower soil bulk density, and provide a slowly available source of nutrients. Studies were conducted to test the hypothesis that an undecomposed material such as Fluff is beneficial as an organic soil amendment that can aid in the establishment of native grasses. While many similarities exist between the land application of other agricultural and industrial waste products such as poultry litters, animal manures (Karlen et al., 1998), and composted biosolids, Fluff is a unique byproduct which required experimental studies to understand the impacts to vegetative establishment, plant nutrient status and impacts to soil quality.

4.1 Land application and vegetation establishment

As previously noted, a potential problem with non-composted organic material is the high C:N ratio, which could create a soil environment with low N availability. However, the creation of low N availability may be an advantage for establishing native vegetation that is adapted to nutrient limited soils and would benefit greatly from a reduction in weed competition for N (Paschke et al., 2000; Barbour et al., 1999; Wilson and Gerry, 1995; McLendon and Redente, 1992). Perennial warm season grasses, such as those native to the Tallgrass Prairie of North America, are well adapted to harsh environmental conditions, including low N availability, giving them a competitive advantage in poor soils (Jung et al., 1988; Wilson and Gerry, 1995; Skeel and Gibson, 1996; Levy et al., 1999). These grasses are used abundantly in reclamation, as they develop extensive root systems that penetrate deep into soils, providing a very effective safeguard against erosion (Drake, 1983). Although these species are highly suited to conservation planting, establishment is a significant barrier to successful utilization, as weedy species can easily overtake them and cause failure, especially in N rich soils (Launchbaugh, 1962; Wedin and Tilman, 1993, 1996; Munshower, 1994; Warnes and Newell, 1998; Reever and Seastedt, 1999; Brejda, 2000).

Studies have been conducted to evaluate the use of Fluff as a soil amendment to successfully rehabilitate damaged military training lands, which often lack sufficient topsoil, organic matter, and nutrients required for successful rehabilitation (Busby et al., 2006; Busby et al., 2010). Busby et al. (2010) carried out a field study in North-Central Tennessee at the Fort Campbell Military Reservation, on an abandoned hay field currently used for Army training activities. Soil at the site was a Sengtown silt loam (fine, mixed, semiactive, thermic, Typic Paleudalfs) (Soil Survey of Montgomery County, Tennessee, 1975). Application of Fluff was made at rates varying from 0 to 36 Mg ha^{-1}. Three warm season grasses species (Big Bluestem - *Andropogon gerardii* , Switchgrass - *Panicum virgatum*, and Indiangrass - *Sorghastrum nutans*) and one cool season grass (Virginia Wildrye - *Elymus virginicus*) were planted. In a separate study, two sites on Fort Benning Military Reservation, GA, were established. The sites chosen were designated as "Dove Field" [a moderately degraded Troup sandy loam soil] and as "Borrow Pit" [highly degraded Borrow Pit soil (highly disturbed Orangeburg Fine-loamy soil (Soil Survey of Muscogee County, Georgia, 1983) (Fig. 2). At these sites, treatment plots consisted of a control where nothing was done, a control with revegetation only, and application of Fluff at rates varying from 0 to143 Mg ha^{-1} with revegetation. As in Tennessee, native grasses Big Bluestem, Switchgrass, Indiangrass, and Virginia Wildrye were planted. Vegetation sampling, including plant biomass (Bonham, 1983), plant nutrient composition, plant species composition (Sharrow and Tober, 1979) and basal vegetative cover, were measured at the end of each of two growing seasons. Plant biomass was collected, consisting of composite samples of all species present. Analysis was performed for total Carbon (C), nitrogen (N), aluminum (Al), boron (B), barium (Ba), calcium (Ca), cobalt (Co), chromium (Cr), copper (Cu), iron (Fe), potassium (K), magnesium (Mg), manganese (Mn), sodium (Na), phosphorus (P), lead (Pb), silicon (Si), and zinc (Zn).

4.2 Vegetation growth and composition

At the Fort Campbell experimental sites, vegetation consisted primarily of agricultural grasses and forbs typical of early successional communities with 49 species in 24 families recorded for the entire study area. After two growing seasons following Fluff application and seeding with the desired warm and cool season grasses, total basal vegetative cover differences were not significant across years or treatments. Annual grass and total annual cover were relatively unaffected by Fluff treatment but were significantly higher in the unseeded control treatment than in the 36 Mg ha^{-1} treatment. The 36 Mg ha^{-1} treatment had significantly higher total perennial, perennial grass, and planted grass cover than both the seeded and unseeded controls. Big bluestem, indiangrass, and switchgrass cover were unaffected by treatment.

Based on the results of the species composition and basal cover analyses, establishment of 2 of the 3 native warm season prairie grasses was enhanced with pulp application rates of 36 Mg ha^{-1}. Indiangrass appears to be relatively unresponsive to the Fluff, but switchgrass and big bluestem showed notable density and cover increases at the highest application rate.

Fig. 2. Borrow Pit field sites on Fort Benning Military Reservation, GA; A) initial application of Fluff, B) plant growth after 2 years.

No differences in biomass were found between the Fluff treatments and seeded control. The lack of change in biomass in the unseeded control plots compared to the seeded plots was most likely due to dominance by ruderal species in the unseeded control plots that typically lack the biomass found in the seeded perennial grasses. Because annual grass cover remained constant but its relative percent composition decreased, it can be concluded that the Fluff was in some way beneficial to the prairie grasses but not inhibitory to weedy species during the first 2 growing seasons following application of up to 36 Mg ha^{-1}. This would be expected for sites with relatively good soil fertility such as those seen at the Fort Campbell experimental sites.

At Fort Benning, a total of 21 species were sampled in the research plots over 2 years. Combined, planted grass species comprised 98.2% of the total species composition of the Borrow Pit and 87.3% of the Dove Field. Application rate had no effect on percent composition of total planted grasses at either site. Switchgrass appeared to be the best suited species as it dominated all seeded sites and comprised the highest relative percentage composition and basal cover of all species present (Fig. 2). It also responded most favorably to Fluff application as basal cover increased significantly with increasing application rate at both sites. Additionally, switchgrass performed so well that the majority of plants produced seed during the first growing season at both sites, which may have contributed to increased dominance the following year. Big bluestem appeared to be unaffected by application rate at the Dove Field site, but basal cover increased significantly with increasing application rate at the Borrow Pit. Given that the more fertile Dove Field site was more conducive to vegetation establishment than the Borrow Pit, this may have been the result of oversupplying nutrients at high application rates which big bluestem was not able to fully exploit at the Dove Field. However, higher application rates overcame deficiencies and created more favorable growing conditions at the Borrow Pit which positively influenced big bluestem growth.

Indiangrass initially performed well in the Dove Field, but remained only a minor vegetation component at the Borrow Pit. Given that indiangrass diminished over time and in response to increased Fluff, while the other two dominant species increased, it appears that indiangrass was not able to effectively compete with switchgrass and big bluestem at either site in the presence of Fluff amended soil. Indiangrass high relative composition in the controls indicates that it was competitive in unamended soils, but its low relative composition in the higher application rates indicates that it was not able to effectively exploit any benefits provided by the amended soils in the manner observed by switchgrass. Further, because it was so much more prevalent in the Dove Field than in the Borrow Pit, indiangrass was not as tolerant to the highly unfavorable growing conditions in the Borrow Pit as were the other species.

Biomass was much higher in the Dove Field than in the Borrow Pit across application rates, but both sites responded very well to increased Fluff application (Table 7). In the Dove Field, biomass remained relatively constant in the unseeded control at less than 300 g m^{-2} but almost doubled in the 143 Mg ha^{-1} treatment from 539 to 1059 g m^{-2} from 2003 to 2004. In the Borrow Pit, the unseeded control lacked any biomass throughout the study, but the 143 Mg ha^{-1} treatment increased from 345 to 582 g m^{-2} over time.

Fluff Rate	Dove Field		Borrow Pit	
	2003	2004	2003	2004
Mg ha^{-1}	------------------- (g m^{-2}) ----------------------			
Unseeded Control	243	291	0	0
0	269	392	18	14
18	344	617	46	90
64	428	613	73	122
72	468	749	202	403
143	539	1059	345	582

Table 7. Biomass yields as affected by Fluff application for the Dove Field and Borrow Pit study sites in 2003 and 2004.

Weedy annual grasses were not affected at the level originally hypothesized. It was expected that annual weeds, with characteristic shallow root systems and intolerance to shading, would respond negatively to increased competition with taller, deeper rooted perennial prairie grasses. Even though annual grasses were unaffected, the increases in switchgrass and big bluestem cover show a positive result of pulp application. Because the planted grass species constituted almost all vegetation that was sampled in the seeded plots (98% in the Borrow Pit and 87% in the Dove Field) and resulted in mean basal cover values of 7.5% and 12.2%, respectively, establishment of a native grass community was considered successful at both sites.

4.3 Plant chemical analysis

Plant chemical composition was also measured to monitor potential changes in plant uptake patterns due to Fluff additions. The measurements were made not only to determine potential changes in the plant health by measuring plant nutrient concentration, but also to measure the potential for environmental concerns with the uptake of heavy metals. In the silt loam soils in Tennessee, soil concentrations of many metals and nutrients were unaffected by Fluff addition, but plant P and Pb accumulation was increased by the 36 Mg ha^{-1} treatment. However, the increase in Pb was insignificant (1.5 mg kg^{-1} for the highest Fluff rate) with respect to established regulatory limits. The increase in soil P concentrations in the high pulp rates alleviated an apparent P deficiency in the study site soils.

Based on these findings, it would be beneficial to use this material as a soil amendment for reestablishing perennial warm-season grasses on disturbed acidic soils with limited P availability. Rates of at least 36 Mg ha^{-1} should be used to achieve noticeable benefits to seeded species, although the upper limit for these benefits has not been determined. The annual limit of Fluff application from a regulatory standpoint based solely on levels of Pb in the material compared to allowable levels in biosolids application would be approximately 230 Mg ha^{-1} $year^{-1}$, with a cumulative limit attained near 4600 Mg ha^{-1}. However, due to logistical challenges and the potentially negative effects on soil physical and chemical properties, these rates would not be advisable. If the highest application rate used in this study were repeated once every five years, the limit would be reached in about 650 years. However, to maintain native grass stands, the annual application rate would be significantly lower due to potential negative compositional changes that could result from nitrogen deposition over time.

In the sandy soils at Ft. Benning, more distinct differences were observed with the increasing rates of Fluff. Plant nutrition was improved at both sites, however, due to very distinctive soils between sites, the effects were dissimilar. At the more productive Dove Field site, plant N, P, K, and Na concentrations increased with increasing Fluff application. At the highly disturbed Borrow Pit site, plant P and Na concentrations also increased with increasing Fluff, as well as Mg concentration. An apparent Fe toxicity problem at the highly degraded site was alleviated by high applications of Fluff, as the control plots and lower application rate treatments accumulated extremely high levels of plant Fe. Plant Ba concentration was also reduced by increasing application of Fluff at both sites. The improved plant nutrition and improvements in cover and biomass of perennial native vegetation at both sites indicates an undecomposed organic material such as Fluff can positively influence the establishment of native vegetation in disturbed soils with highly variable properties. Results indicate that greater benefits are achieved with higher levels of soil degradation when using fluff to aid in establishment of warm-season prairie grasses.

4.4 Impacts on soil

An important consideration in the utilization of Fluff as a soil amendment is what impact it might have on soil condition or quality. To examine the potential impact on soil chemical and physical conditions, soil samples (Prior, et al., 2004) were collected following the application of Fluff on degraded US Army training grounds in both sandy loam and silt loam soils (Torbert et al., 2007; Busby et al., 2010). Soil samples were obtained at depths of 0-5, 5-10, 10-20, and 20-30, 30-60 and 60-90 cm and analyzed for total N and C, nitrate, and ammonia (Nelson and Sommers, 1996). Extractable Al, As, B, Ca, Cd, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, Pb, S, Se, and Zn concentrations as well as soil pH and bulk density were also determined (Bremmer, 1996; Soltanpour et al., 1996; Hue and Evans, 1986).

4.4.1 Silty-loam soils

For silty loam soils, few treatment effects were found for soil nutrients analyzed. Soil C and P concentration was higher with 36 Mg ha^{-1} fluff application than in the unseeded control, but soil N was unaffected by Fluff application. Impacts were also noted for soil K, Ca, Mn, and Cu with Fluff application. Few differences were observed for soil heavy metals, but Fluff application did impact Pb, Al, and As when extracted with Mehlich III extractant (Mehlich, 1984). Mean As concentrations were lower in the Fluff treatments than the unseeded control, and Pb concentration increased approximately 1.5 mg kg^{-1} in the 36 Mg ha^{-1} treatment over the controls.

The analysis of soil chemical properties indicated that Fluff application can significantly increase available P in soils. The increase in extractable soil P in the highest application rates combined with a stable and sufficient level of plant P indicated that an adequate amount of labile P was supplied by Fluff rates greater than 18 Mg ha^{-1} in this silt loam soil. Whether the effect of increased plant P accumulation is a direct result of Fluff supplied P or by some other mechanism is unknown. Because weedy plants usually respond better to fertilization than warm season prairie grasses, this result may have been due to increased mycorrhizal infectivity as weedy grasses did not diminish with increasing application rate, but prairie grasses increased (Noyd et al., 1995, 1996). However, given that soil P levels only increased in the depths where Fluff was incorporated, decomposition of the Fluff and subsequent mineralization of P was most likely responsible. The added P from Fluff may have promoted N immobilization, which would affect annual species more than the planted perennial grasses, as the prairie grasses are much more efficient at nutrient utilization (Brejda, 2000). This would explain why plant shoot P concentration, soil P concentration, cover of planted grasses, and Fluff application rate were all directly related, but soil and shoot N concentrations and annual grass cover were unaffected.

Although soil Pb levels increased significantly from a statistical standpoint in the upper profiles at high Fluff rates, there was no significant change from a regulatory standpoint: amounting to a net increase of approximately 1.5 mg kg^{-1} in the top 30 cm of the soil profile at the highest Fluff application rate. Additionally, both P and Pb only increased in the top 10 cm where the Fluff was incorporated, indicating that no movement into the lower soil profile was occurring after 2 growing seasons. Because Pb is very tightly bound by soil organic matter, it does not readily leach through the soil profile and is largely unavailable for plant uptake (Kabata-Pendias, 2001).

4.4.2 Sandy-loam soils

In sandy loam soils (Torbert et al., 2007), the addition of Fluff had an impact on the soil bulk density level in the surface soil (0-5 cm). While no significant difference was noted for depths below 0-5 cm at either study site, the impact of improving the soil bulk density in the soil surface would be important for soil quality and native grass establishment. At the Dove

Field, the soil bulk density was in the range of 1.56 g cm^{-3} at the initiation of the study, but with the application of 143 Mg ha^{-1} Fluff, soil bulk density was drastically reduced to 1.17 g cm^{-3}. An even larger impact was observed with the soil at the Borrow Pit site, where the initial level of soil bulk density was 1.83 g cm^{-3}. The addition of Fluff at this site reduced the soil bulk density to 1.22 g cm^{-3} with application rates of 143 Mg ha^{-1}.

The level of reduction observed with Fluff application would have an important impact on soil condition at both locations. Soil bulk densities above 1.5 g cm^{-3} have generally been shown to be detrimental to root growth and plant yield (Gliski and Lipiec, 1990). The reduction in the level of bulk density observed in this first year would be much more conducive to both plant establishment and root growth of the native grasses. The soil bulk density levels observed from second year soil sampling indicated that the soil physical condition had been substantially improved and that this improvement would likely persist. The improvement in soil bulk density alone would indicate that the degraded soil conditions commonly associated with US Army training activities could be substantially ameliorated with high Fluff application rates.

The ability of the soil to provide plant nutrients is controlled by many factors, such as organic matter content, soil pH, and soil texture (Potash and Phosphate Inst., 2003; Mengel and Kirkby, 1982). Many of these factors, such as soil organic matter content, are reduced in degraded soils, thereby reducing the ability of the soil to provide adequate plant nutrient supply. As noted, the Fluff contained substantial amounts of essential plant nutrients, which would have been present with the application of the Fluff (Table 1). However, these nutrients would not necessarily be available for plant uptake, depending on the condition of the soil, particularly the soil pH level, and the decomposition and release of the nutrients in the Fluff (Potash and Phosphate Inst., 2003).

Extractable soil nutrients (Mehlich, 1984), measured at the end of the first growing season for both sites, are shown in Table 8**.** The application of Fluff increased extractable nutrients in the surface soil layer at both sites. At the Dove Field, a less degraded soil compared to the Borrow Pit, Fluff application resulted in a significant impact on P, B, Ca, Co, and Zn. The soil concentration of Ca and P were particularly improved with the application of Fluff, with Ca concentrations increasing from 195 to 1835 mg kg^{-1} and P concentrations increasing from 29 to 145 mg kg^{-1} with the application of 143 Mg ha^{-1} of Fluff. The concentration of extractable P in soil often limits plant production in agricultural scenarios, which results in the need to add P fertilizer to improve soil fertility (Potash and Phosphate Inst., 2003).

At the Borrow Pit, the soil was extremely degraded, resulting in almost no vegetation at the site at the start of the study and the initial soil fertility level being extremely low. The application of Fluff resulted in a significant increase in the extractable soil nutrients B, Ca, Co, Cu, Fe, K, Mg, Mn, P, and Zn (Table 8). This increase was likely due not only to the addition of these nutrients with the Fluff, but also due to the improvement in the soil pH level that was observed with increasing levels of Fluff application. As soil pH level increases toward neutral, the availability of most plant nutrients improves (Potash and Phosphate Inst., 2003). The addition of Fluff increased the soil extractable levels of plant macro- and micro-nutrients to levels that would allow adequate plant growth.

Soil extracts were also analyzed for concentration of the heavy metals Cd, Cr, Ni, and Pb (Table 9), which have USEPA limits for biosolids application (U.S. Government 40 C.F.R. Part 503, 1999). The concentration of Cd was increased with increasing Fluff application and Pb increased as well, but only at the highest application rate. The concentration of Cr, Ni, and Pb were also increased, but only at the highest application rate. None of the heavy metal concentrations found in the soil would be of concern in terms of the maximum cumulative loading limits as regulated for biosolids (U.S. Government 40 C.F.R. Part 503, 1999).

Fluff rate	P	K	Ca	Mg	Mn	Fe	Zn	B	Cu	Co	Na
Mg ha^{-1}	-- (mg kg^{-1}) --										
					Dove Field						
0	29.7	53.5	225	59.6	21.6	11.6	1.56	0.05	0.32	0.08	6.3
18	58.3	57.4	572	79.0	28.1	14.0	6.80	0.27	0.54	0.13	13.0
64	64.2	53.0	745	46.9	25.8	15.3	8.52	0.11	0.72	0.14	9.1
72	66.0	66.6	663	44.8	33.0	14.6	9.72	0.16	1.53	0.14	9.3
143	145	86.5	1835	79.2	33.3	16.3	25.4	0.54	1.67	0.17	19.4
					Borrow Pit						
0	2.02	9.1	25	2.5	1.0	3.9	0.9	0.01	0.14	0.01	8.1
18	12.0	11.8	194	7.1	1.5	5.6	2.8	0.08	0.31	0.02	12.1
64	5.5	19.0	101	7.6	1.7	5.6	1.7	0.05	0.59	0.02	9.7
72	65.7	24.6	835	18.8	6.0	12.4	17.0	0.23	2.06	0.05	15.2
143	102	36.8	1511	41.0	8.6	23.3	19.7	0.71	2.42	0.10	93.9

Table 8. Soil extractable plant nutrient concentrations in the 0-5 cm soil depth for the Dove field and Borrow Pit study sites.

The application of the Fluff had a large impact on the soil pH, especially in the soil sampled after the first growing season. The Fluff would not be a liming material (McLean, 1982), but because of the near neutral pH and large Ca content of the Fluff material, the application of Fluff raised the soil pH. In the first year of the study, soil pH had a linear response to increasing Fluff application at both study sites. This increase in soil pH could be critical to the establishment of native grasses. Soil pH at or below the 5.3 level would be very detrimental to plant growth, resulting in nutrient deficiencies and potential Al toxicity (Potash and Phosphate Inst., 2003). The level of soil pH observed in the control plots would partially explain the complete failure of plant growth that was observed in the Borrow Pit site.

Fluff rate	Ba	Cd	Cr	Ni	Pb
Mg ha^{-1}	-------------------------- (mg kg^{-1}) --------------------------				
		Dove Field			
0	0.63	0.05	0.03	0.08	0.00
18	0.47	0.12	0.11	0.16	0.27
64	0.45	0.08	0.11	0.45	0.03
72	0.45	0.10	0.11	0.22	0.02
143	0.52	0.21	0.28	0.50	0.80
		Borrow Pit			
0	0.47	0.01	0.01	0.02	0.15
18	0.54	0.01	0.04	0.10	0.31
64	0.75	0.01	0.02	0.05	0.21
72	1.04	0.07	0.14	0.31	0.87
143	1.97	0.13	0.35	0.77	2.26

Table 9. Soil extractable heavy metal concentrations in the 0-5 cm soil depth for the Dove Field and Borrow Pit study sites.

Soil C and N concentration was measured at both study sites. Soil C and N concentration is one of the most important factors for assessing soil quality (Wienhold et al., 2004) that impacts soil physical, chemical, and biological functions of the soil. The buildup of soil C can be essential to the long term health of the soil system.

At the Dove Field, in plots where no Fluff was applied, soil C concentration was approximately 13 g kg $^{-1}$ in the surface 0-5 cm depth and declined with increasing soil depth, down to 3.3 g kg^{-1} at the 30-60 cm soil depth layer. Soil N concentration was found to be 0.6 g kg^{-1} in the soil surface (0-5 cm) and fell to 0.2 g kg^{-1} at the 30-60 cm soil depth layer. These levels of soil C and N are in the range expected for degraded sandy loam soils in the region. The application of Fluff had a large impact on the soil concentration of C in the soil surface (0-5 cm), increasing with increasing Fluff application up to approximately 39 g kg $^{-1}$ (Fig. 3). Likewise, a significant linear regression was observed for soil N, increasing with increasing Fluff application rate **(**Fig. 3). No significant impact from the application of Fluff was observed for soil concentration of C and N below the 0-5 cm depth at this location.

In the highly degraded Borrow Pit site, the soil concentrations of C and N were extremely low where no Fluff had been applied, with a C concentration of 2.2 g kg^{-1} and N concentration of 0.1 g kg^{-1}. Interestingly, little difference was observed throughout the entire soil profile for C and N concentration, as reflected by the extremely low concentrations and the lack of any plant growth. However, the application of Fluff resulted in a significant influence on soil C in the surface 0-5 cm depth increment, with an increase to approximately 20.2 g kg^{-1} for the 143 Mg ha^{-1} Fluff application rate (Fig. 4). Likewise, the soil N level was increased with increasing Fluff application, to approximately 1.0 g N kg^{-1} with the 143 Mg ha^{-1} application rate. This level of increase in soil C and N at this depth demonstrated an improvement in soil condition and is in the range that would be considered excellent for a sandy loam soil in this region.

Unlike the Dove Field soil, significant linear regression was observed for increasing soil C and N with increasing Fluff application below the 0-5 cm depth (Fig. 4). While small compared to the impact that was observed in the 0-5 cm depth, a distinct increase in both C and N concentration could be observed with the increasing application of Fluff at the 5-10, 10-20, and 20-30 cm depth increments. This increase could be partially caused by the movement of soluble C and N compounds deeper into the soil profile. However, this increase was most likely the result of increased plant rooting with the establishment of the native grasses. The increased grass biomass observed with increased Fluff application rate would have been accompanied by increased root biomass below the soil surface resulting in increased organic matter input into the soil. This improvement in soil C and N not only at the soil surface where Fluff was incorporated, but deeper into the soil profile would be invaluable to improving the soil/plant environment on a highly disturbed soil.

The results of this study indicated that the application of a non-composted organic amendment to highly acidic, degraded soils would improve soil conditions and provide a healthier soil environment for plant establishment. The improved conditions were most prominent on the more highly degraded soil, indicating that the more degraded the soil the higher the potential benefit from the addition of organic amendments (even non-composted organic amendments).

4.5 Dust control

The organic byproduct of the WastAway Garbage Recycling System has proven effective as a soil amendment to reestablish native grasses following disturbance on installation training

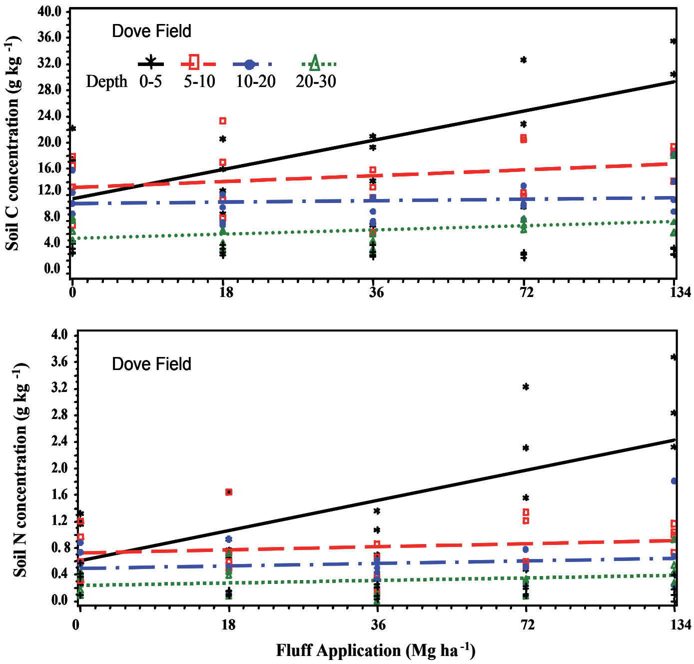

Fig. 3. Regression relationships of Fluff application rate to soil C and N concentration measured at 0-5 5-10, 10-20, and 20-30 cm soil depth at the Field study site in 2004.

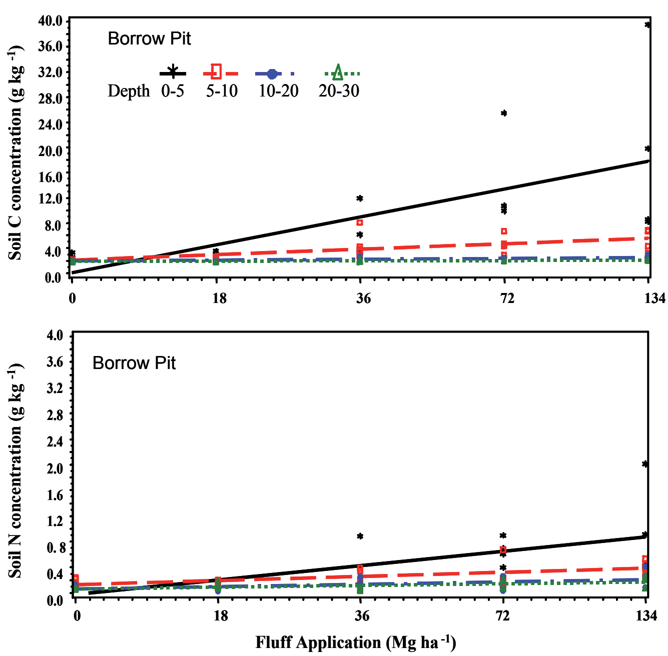

Fig. 4. Regression relationships of Fluff application rate to soil C and N concentration measured at 0-5 5-10, 10-20, and 20-30 cm soil depth at the Borrow Pit study site in 2004.

lands. Because this material is derived from the organic component of household waste, a major portion of which is cellulose, it has many peculiar properties offering potential utilization in many different scenarios, including dust suppression.

Cellulose is the most abundant carbohydrate on Earth and one of the most intensively studied organic compounds, due to its universal importance in fiber and polymer production, paper products, and numerous other industrial applications. Lignosulfonate, a paper processing byproduct, has been extensively used by Departments of Transportation in the southwestern United States and the forestry industry in the western and southeastern United States for dust control on unsurfaced county and logging roads (Gebhart and Hale, 1996). Because of the high lignin and cellulose content of Fluff, it shares similar dust control properties with commercially produced lignosulfonates. Additionally, the textural characteristics and pore space of Fluff make it an ideal candidate for use as a dust control agent alone and in combination with other dust control compounds such as vegetable oil and calcium chloride which have been used in this capacity for decades around the world (Gebhart et al., 1999).

In June of 2006, a series of field tests were conducted near McMinnville, TN, to evaluate the performance of Fluff, alone and in combination with vegetable (soybean) oil and calcium chloride. Three unsurfaced test roads were selected and divided into three segments, each of which randomly received one of the following treatments: Untreated control; Fluff alone at a rate of 35.8 Mg/ha; Fluff plus vegetable oil (100 ml/kg Fluff); and Fluff plus 38% Calcium chloride flake (10g/kg Fluff). Following treatment application, each road segment was subjected to routine local traffic for a period of 100 days to evaluate dust control efficiency through time.

At about 50 day intervals, each road segment was subjected to controlled traffic using a vehicle equipped with a mobile dust plume monitor to determine an emission index for segments of a given test road. The method chosen to determine the emission index was mobile monitoring of the PM-10 concentration in a representative part of the dust plume generated by a test vehicle on the unpaved road. A DustTRAK model 8520 was used for this purpose, with one second concentration measurements. The inlet to the DustTRAK sampling line was secured along the side of the test vehicle, thereby sampling the dust plume from the right front tire. The inlet was placed midway between the front and rear tires of the test vehicle, thereby avoiding potentially large fluctuations in the plume concentration due to the wake of the vehicle.

Emissions testing began from a stationary position at the beginning of each test segment and accelerated to 35 kph for travel and sampling through each segment. Each test provided nine DustTRAK data runs per test road. Time markers were determined for the DustTRAK output so that the reference points on the treated road segments could be correlated with the DustTRAK measurement datalog.

Table 10 shows average PM-10 concentrations for each of the dust control treatments on two dates for the three test roads. For each test road, the Fluff plus vegetable oil treatment was found to be the most effective dust control treatment, followed by Fluff plus Calcium Chloride, Fluff alone, and lastly, the untreated control. During the September 2006 testing, emission rates were substantially reduced for all test roads because of recent rains and high moisture content of the road surfaces. Nevertheless, the treated segments still showed moderate to high levels of control efficiency when compared to untreated segments, indicating that Fluff, whether alone or in combination with other dust control compounds, has the potential for low-cost, long-lasting dust control on moderately traveled unpaved roads. Given its proven potential as a soil amendment, this additional use of Fluff demonstrates yet another beneficial reuse of this municipal solid waste processing byproduct.

Road	Sample Date	PM-10 Concentration (mg/m3)			
		Fluff/Oil	Fluff/CaCl2	Fluff	Untreated
1	7/18/06	0.85	12.26	14.36	107.19
	9/20/06	0.07	0.07	0.15	0.83
2	7/18/06	0.61	3.73	5.73	56.91
	9/20/06	0.29	0.63	0.81	5.71
3	7/18/06	0.50	3.49	6.66	16.84
	9/20/06	0.31	1.01	1.22	2.44

Table 10. Average PM-10 concentration for each dust control treatment measured on two sampling dates for unpaved test roads near McMinnville, TN.

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12

Waste to Energy, Wasting Resources and Livelihoods

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1. Introduction

Not recovering the material embedded in solid waste means wasting resources and thus reinforcing the pressure to further extract natural resources for the manufacturing of new products. Industrial ecology, life cycle analysis, material flow analysis, ecological footprint and other approaches and concepts have long ago already demonstrated the necessity and possibilities of reintegrating recyclable materials into production flows, reducing the waste of resources and thus sparing the environment. Far too often however, business is done as usual and the status quo of production and consumption is not altered significantly.

The prevailing perceptions of waste are still based on the understanding that waste is something worthless, unused or has ceased to be useful for human purpose. The word waste comes from the Latin *vastus*, meaning unoccupied or desolate and is akin to the Latin *vanus* (empty or vain) (Lynch, 1990). Originally waste meant something useless and hostile to humans, to be ignored and discarded. Products and packaging usually have a defined lifespan. Sometimes the product life is shortened for the purpose of inducing larger consumption rates. Nor producer, nor consumer are generally concerned about the final destination of these materials. However, with reuse and recycling these materials again become potential resources. Legislation implementing reverse logistics has the potential to alter the established wasteful cycles. Incineration to recover the energy from waste can not be considered a sustainable recycling practice, since it is not an energy efficient process and once burnt the resources are gone for ever.

The statistics evidence that we live in a time of waste explosion. Never has humanity generated so much refuse during production and garbage after consumption as in current times. It is estimated, for example, that globally, 20–50 million tons of E-Waste, the newest category of waste, which includes electronic and electric equipment, are discarded annually (Ongondoa et al., 2011). The authors confirm that the penetration of electronic equipment in a number of countries in the global South is approaching the level of industrialized countries. In Brazil the increasing generation of E-Waste is becoming a noticeable concern. Most of this waste comes from obsolete mobile phones, telephones, TVs, computers, radios, washing machines, refrigerators and freezers. In 2006, the per capita E-Waste rate in Brazil already stood at 2.6 kg, compared to the global average of 1 kg/per person/per year (Rocha, 2009).

Even remote rural towns, almost everywhere, have to deal with increasing generation of waste and growing complexity of the waste composition. At the same time household garbage has become more industrialized, more toxic and less biodegradable. With the advent of globalised mass consumption, coupled with the lack of adequate spaces to discard these materials, particularly in city regions, Governments, producers and consumers are under pressure to find adequate solutions to the problems created by solid waste.

In the forefront of the current waste management debate is the promotion of new technologies for waste treatment. Less attention is given to considerations that suggest resource economy, reuse, recycling and changes in production, consumption and lifestyles to generate less waste at first. Recommendations that question the continuous, growth oriented economic development and consumption patterns are less popular and usually silenced in order to maintain the status quo. Marxist perspectives underline the fact that capitalism requires a steady acceleration of wasting, discarding and abandonment, in order to keep a scarcity of goods. Scarcity coupled with an artificial inflation of consumer desires, increases the throughput of material in our system, and thus maintains the rate of profit in the face of its progressive tendency to fall.

Solid waste incineration is propagated by business and the media as an efficient management solution, because of the rapid handling of the discarded materials, the diminished need for new landfills and the generation of energy as a by-product. Yet, the environmental and social dimensions of this technological approach to waste often remain unconsidered. Social and environmental injustice may arise from locating these technologies and from displacing the workers who already make a living through resource recovery. Deliberating authorities often overlook the wider implications from deviating recyclable materials away from the recycling sector.

This chapter will analyze the recent emergence of 'waste for energy' (WfE) proposals in Brazil. The discussion will consider particularly the social perspectives related to waste management decisions, looking at existing informal and organized recycling schemes. The government supported selective waste collection and recycling initiatives in the cities of Diadema and Londrina will showcase viable solutions in integrated waste management. Expensive 'waste to energy' schemes are considered unsustainable for generating environmental harm and for perpetuating the waste of natural and human resources.

1.1 Trends in household waste generation

"People consume leisure, space and time as if our lives were simply an eating up and a throwing away […] it is clear that capitalism, once it is connected to the mass market, is motivated to increase consumption" (Lynch, 1990, p. 148).

Unsustainable lifestyles have permitted and motivated ruthless natural resource extraction with disastrous results for the environment, and in particular for indigenous and traditional communities. Media reports on new environmental and social impacts from mining, fishing, forestry, cattle ranching, industrial activities, transportation, tourism, etc. reach us every day through Internet, radio, television, theatre, art, film, music and written sources. The links between resource over-exploitation and environmental disasters (culminating in climate change) seem direct and clear and yet are ignored or denied. In fact, most societies consume more resources than a sustainable living would allow. The prevailing western economic development model has allowed for unprecedented accumulation of wealth while the number of socially and economically excluded people continues on the rise. Naomi Klein evidences these perverse facets of economic growth based on the exploitation of nature and society in her book '*The shock doctrine*' (Klein, 2008). The price we pay in terms of losses in biodiversity and cultural diversity is high, just to maintain, further disseminate and accelerate the status quo of mass consumption and unsustainable lifestyles. The problems generated by increasing waste quantities are ubiquitous.

The quantity of solid waste, in Europe and North America in particular, has increased in close relation to economic growth, over the past decades, attested by the growing solid waste quantities along with increases in Gross Domestic Products (GDPs). A Swedish study from Sjöström and Östblom (2010), for example, mentions a total quantity of municipal waste per capita increase of 29% in North America, 35% in OECD countries, and 54% in the EU15 between 1980 and 2005.

Packaging magnifies the task of household disposal because of its bulky proportions and its mixture with decomposable garbage. For the sake of convenience and the prevention of spoilage and disease products are wrapped more than ever, often using materials, which do not decompose, are toxic, or are still difficult to recycle.

Although household waste manifests only a fraction of the solid waste generated, its reduction can be key in promoting a paradigm shift towards more sustainable production and consumption patterns. Construction waste, industrial waste, mining waste, and agricultural waste are also linked to consumption and lifestyles. In 2005, the UK produced approximately 46.4 million tons of household and similar waste with 60% of this landfilled, 34% recycled and 6% incinerated. Only 11% of the estimated waste was household waste, compared to 36% construction and demolition, 28% mining and quarrying, 10% industrial, 13% commercial waste, and less than 1% agricultural and sewage waste (Department for Environment, Food and Rural Affairs [DEFRA], 2006).

Despite the prevailing waste of resources, there are also initiatives concerned with the reduction and ultimately the generation of *zero waste*. Banning plastic bags is often one of the first actions promoted by local governments and some business towards reducing plastic waste and, although important, only targets the tip of the iceberg. Lifestyle changes suggested under the *voluntary simplicity* initiative are perceived as another form of individuals impacting these developments. These measures are all important, however they need to come together with policy instruments in order to reduce waste intensities and to alter the final destination of waste.

1.2 Trends in municipal solid waste management

Although worldwide landfilling is on average still the most widespread form of waste disposal, more and more cities are moving away from waste deposits towards recycling and incineration. In India almost 90% of the collected household waste is still deposited at uncontrolled sites (Talyan et al., 2008). In Turkey too, dumping solid waste on open sites is still the prevailing method, followed by sanitary landfills (Agdag, 2009; Turan et al., 2009). The final destination in the United Kingdom, Canada and the United States for over 50% of the household waste is still the controlled landfill, however here too the trend goes towards increased recycling. Sweden is one of the few countries, which already has a reduced percentage of waste disposed at landfills; and it is also one of the countries with the highest waste incineration rate (Persson, 2006).

Less generation of waste, more material recovery, energy from waste and much less landfills seems to be the guiding principles in many European countries (DEFRA, 2007). Within recent decades, one of the major arguments for waste incineration in the global North has been the energy generation from solid waste and the potential fossil fuel saving. The following table summarizes some country's waste incineration capacities (Table 1).

Similar developments are occurring in North America. In the US for example already 12.6% of the household waste was incinerated in 2007 (Vyhnak, 2008). Japan, South Korea, Taiwan and Singapore are the Asian countries with the largest number of incinerators (Gohlke & Martin, 2007; Bai & Sutanto, 2002). In Latin America the number of incinerators is still small and addresses mainly hospital and industrial waste. In the 1970s and early 1980s municipal governments in São Paulo and Buenos Aires had contemplated the expansion of incinerators for household waste, however, at that time social mobilization and the high cost of this technology prevented its establishment. Waste incineration has now re-emerged in Brazil and in other countries in Latin America as 'waste for energy' plants.

Country	Number of establishments	Tons/year
Holland	11	488,000
UK	19	266,000
Sweden	31	136,000
France	210	132,000
Italy	32	91,000

Table 1. WfE establishments in some European countries. Source: Longden et al., 2007; European Environmental Agency [EEA], 2009).

How do cities in Brazil cope with the rapidly mounting quantities of discarded material? In Brazil 25.5% of the municipalities still dump their waste on uncontrolled landfills, while another 19.6% deposits the waste on controlled landfills (Associação Brasileira de Empresas de Limpeza Pública e Resíduos Especiais [ABRELPE], 2007). Officially the recycling rate in Brazil is still insignificant, with approximately 2% of the waste being recovered through government supported selective waste collection programs (Brazil, 2009). Throughout Brazil, as well as in other Latin American and Asian countries there are numerous experiences where organized recycling groups engage at different levels with Government in order to perform selective waste collection in their city. In many cases the recyclers have already established a history in the community with door-to-door collection and partnerships with business and industry. It is important to note that the official number for recycling does not include the effort of tens of thousands of informal recyclers working throughout Brazil, as well as in most other countries in the global South. In Brazil, for example, there are between 800,000 to one million informal and organized recyclers (called *catadores*), according to the national recyclers movement (Movimento Nacional de Catadores de Materiais Recicláveis [MNCR], 2010). These people make a livelihood from resource recovery, contribute to resource savings, and diminish environmental hazards by redirecting the materials.

Uncontrolled landfills, such as the famous *Gramacho* landfill in the metropolitan region of Rio de Janeiro, recently portrayed in the award winning movie *Waste land* and in the documentary '*Beyond Gramacho*', are still a reality in some parts of Brazil. With the implantation of the recently approved federal solid waste management law (Law Nº12.305/2010 - *Política Nacional de Resíduos Sólidos*), however, the days of uncontrolled landfills are counted until 2014, when all uncontrolled waste dumps need to be eliminated and every city is required to have their waste management plan in place.

Given the pressure on municipalities to finding adequate forms of waste management, many governments perceive incineration as a quick and simple alternative. Thermal and bio-mechanic treatment of waste is gaining momentum in many parts of Brazil, as municipalities in Latin America and Asia are being offered expensive *Waste for Energy* technology as a solution to their waste crisis.

2. Social and economic reflections on *Waste for Energy* (WfE)

This section introduces social, environmental, and philosophical questions related to *Waste for Energy*, without detailing the technical aspects of the various technologies. As discussed earlier there is a tendency in Europe and North America to set up waste for energy plants, supported by specific funding programs and converging energy and waste legislation. In England for example the *Energy White Paper* (Department of Trade and Industry, 2007) and the *Waste Strategy for England* (DEFRA, 2007), advocate for waste being a resource to generate biomass fuel as well as heat and power. "Energy from waste is expected to account for 25% of municipal waste by 2020 compared to 10% today" (DEFRA, 2007, p. 7). There have already been a number of inter related projects that have facilitated investment in renewable energy and waste infrastructure.

To transform solid waste into energy is an attractive proposal, given the pressure put on governments in terms of achieving greater shares of energy from renewable sources. For example, the EU's target to achieve alternative energy supply is at 20% by 2020. Increased recovery of energy from waste is interpreted as a key objective to help reduce greenhouse gas emissions by diverting greater amounts of biodegradable waste away from landfills and by increasing the recovery of energy from waste. In the EU governments have promoted measures to stimulate energy recovery from solid waste. Such measures include the "banding of the Renewables Obligation ('RO'), extending enhanced Capital Allowances ('ECAs') to include Solid Recovered Fuel ('SRF') related equipment along with a heightened expectation for energy generated from waste management activity to achieve the most climate change friendly outcome through the use of 'CHP' [combined heat and power]" (DEFRA et al., 2009, p. 4).

Recent technology developments see solid waste converted into recovered fuel pellets. These would, for example, be produced locally and transported to large-scale gasification and petrochemical facilities to be used in substitution for diesel or gasoline fuel. The European oil and automotive industries are supportive of WfE technology as a means to meet the current and future bio-fuel directive. Solid waste recovered fuel is "*prepared from non-hazardous waste to be utilised for energy recovery in incineration or co-incineration plants…*" (DEFRA et al., 2009, p. 9). The critique from environmentalists is usually related to climate change impacts with carbon dioxide generation from these plants and the high costs for this technology. These expenses could be invested in more environmentally sound and climate friendly energy, tackling the problem at the roots.

The examples on energy policy supporting the use of solid waste as 'alternative' fuel in the UK, are representative for the trend in many countries in Europe and North America. Rising prices for fossil fuel over the past decade are often mentioned to justify WfE. Waste fuels are eligible for revenues under the *Renewables Obligation* and the *EU Emissions Trading Scheme*. Existing protocols and standards for the use of waste fuels are adjusted to facilitate the options provided by WfE. Particularly climate change and renewable energies legislation consider WfE technology a legitimate form to be funded under *Carbon gaining funds.*

Industry has addressed the negative image that is attached to waste incineration by referring to the technology primarily as energy recovery form. The following quote highlights a dominant engineering perspective, failing to understand the larger environmental and social picture. "Waste should be regarded as a fuel rather than something which needs to be treated - Unfortunately, most legislation over recent years has erroneously and dogmatically focused on WfE as waste treatment rather than as energy production, and has attempted to deal with an WfE plant as if it were an incinerator, rather than a power station" (Institution of Mechanical Engineers, n.d., p. 18).

Waste management decisions often favour incineration as a quick and efficient solution and governments assist the process of obtaining local planning consent and licensing implantation for WfE plants as power plants. In addition, there are many other drivers for WfE, including:

- Increasing costs of WfE treatment (and disposal),
- rising energy demands,
- potential to quickly reduce the large volume of waste generated daily,
- understanding that energy can be generated from waste and converted into electricity, erroneously promoted as "green energy",
- reduced costs with workforce,
- potential to receive government revenue or to avoid costs from the use of waste fuels.

The trends observed in countries in the global North are making its way to the countries in the global South. Here the public is usually not well informed about the risks, the costs or alternatives. Multinational concerns and consulting firms approach governments in these countries to showcase the technology and to promote accessible public-private funding schemes for local governments to implement WfE technology. Most often these decision processes happen without ample community awareness and participation.

2.1 Major concerns with Waste for Energy approaches

- *WfE is not a form of recycling*

Solid waste incineration with energy recovery is often referred to as recycling, and is therefore credited with the benefits and the positive image of recycling. However, the term 'recycling' means "*recovery and reprocessing of waste materials for use in new products. The basic phases in recycling are the collection of waste materials, their processing or manufacture into new products, and the purchase of those products, which may then themselves be recycled*" (Britannica Online Encyclopedia, n.d.). Following this rational, solid waste is understood as renewable resources. However, the resource solid waste is only renewable if recycled. Waste to energy makes it a non-renewable resource.

Furthermore, with WfE the need to adopt a materials flow, a cyclical approach is not met. This technology does not involve a cyclical course, since the material dies with incineration. WfE is considered recycling, however, the final product of this industry is energy, which is a final stage, whereas in material recycling any other product can be recycled at least twice.

- *WfE is not a 'green' technology*

WfE is often considered a 'green' technology because it reduces potential methane gas emissions, which would be generated at the landfill. However, the incineration process itself also generates greenhouse gas emissions, despite the claim of being a *Carbon saving mechanism.*

Depending on the material, on the process and the local circumstances, recycling also results in a net reduction of greenhouse gas emissions, however, with the benefit of also reducing emissions related to new resource extractions. Organic waste recycling and composting though benefit the methane gas reduction at landfills.

- *WfE is not energy efficient*

Despite WfE not necessarily being energy efficient, in the UK this technology is considered under the *Renewable Obligation Certificate* (ROC), which is the main support scheme for renewable electricity projects in the UK. It places an obligation on UK suppliers of electricity to source an increasing proportion of their electricity from renewable sources. Ironically WfE falls under this regime. In the UK, combined heat and power plants (CHP) continue to receive 1 ROC/MWh of electricity generated and Biomass CHP plants will receive 2 ROCs/MWh (DEFRA et al., 2009).

- *Growth oriented WfE*

WfE assumes growth in solid waste generation. For example, in the UK the expected increase of 1.5% per year signifies an arising of 37 million tons of waste in the year 2020 (DEFRA, 2007). Again the proposal of WfE is anchored in a growth-oriented paradigm. In order for WfE to be considered economical al and to meet the continuous increased energy demands, there will have to be an ever-increasing amount of solid waste, which is unsustainable.

- *Decisions to implement WfE are usually not participatory*

The need to engage with all stakeholders is not met in the case of the recent expansion of this technology in Brazil. Informal and organized recyclers are major stakeholders in waste management and they are excluded from the decision making process.

2.2 Multinational funding of Waste for Energy

In the early 1990s the trend of the private sector becoming more independent of government agencies and the public sector becoming more businesslike started to become noticeable (Larkin, 1994). Economic globalization has allowed for the private initiative and particularly large corporations to expand into basic infrastructure and service provision, which until then were generally provided through the government. Municipal waste management was one of the last public sectors to become explored by private capital. During the past few years large-scale technologies such as incineration or automatized selective separation plants have massively entered the waste management market, also in the global South.
Public Private Partnerships (PPP) and Private Funding Initiatives (PFI) are common in the funding of these expensive incineration technologies. PPPs are considered an alternative to full privatization and a solution for municipalities to tackle basic infrastructure and service provision related to water, sewage and waste. Rapidly increasing urban population, following consumption-oriented lifestyles, has generated serious disposal problems in most cities in the global South.
Through PPPs "government and private companies assume co-responsibility and co-ownership for the delivery of city services … [and] the advantages of the private sector—dynamism, access to finance, knowledge of technologies, managerial efficiency, and entrepreneurial spirit—are combined with the social responsibility, environmental awareness, local knowledge and job generation concerns of the public sector" (Ahmed & Ali, 2004, p. 471). Ideally this arrangement should improve the efficiency of the entire solid waste management sector. This means, however, that governments can become locked into

long-term contracts, without the necessary control over strategic decisions or over the quality and price of the service. Whether to partnership with recycling coops or whether to employ or subcontract recyclers in waste management might not be an option for local governments that have contracted out the waste management service.

The new federal solid waste regulation puts municipalities under pressure to invest in appropriate solid waste management. Most of the officially collected solid waste in Brazil is still discarded at uncontrolled waste dumps, causing severe environmental health problems. This situation has to change by 2014, according to the new federal law, when cities are required to have an alternative solution in place for the final solid waste destination. On average 4% of the municipal budget in Brazil is currently directed towards solid waste management, a figure that is insufficient to make the necessary radical changes away from waste dumping. Hence, PPPs are considered a solution to overcome the solid waste predicament, as suggested by the Brazilian Solid Waste Management Association (Maximo, 2011). In accordance, the federal government has already made available specific credit lines through the two Brazilian banks, *BNDS* and *Caixa Econômica*, and by way of specific funds to be used by municipalities to upgrade their solid waste management systems.

2.3 Social implications of waste incineration

Although still embryonic, many civil society groups and academics have manifested concerns about WfE technology in many parts of the world. Besides the environmental impacts, with dangerous air pollutants and toxic ashes, waste incineration allows the current unsustainable situation of resource extraction, production, consumption and discarding to be maintained. The switch towards incineration technology does not require the producer or the consumer to change habitual ways of producing and consuming. Once a material is burnt, however, the resource is not renewable any more and will not be able to be used as the same resource. Nevertheless, incineration is advertised as renewable energy, as recycling and even as clean development mechanism. These misconceptions need to be rectified. *Waste to Energy* technologies terminate the possibility of recycling and therefore reiterate new resource extraction.

Furthermore, there are important social considerations to be made. Incineration does not consider those who are already in the business of making different things with and from solid waste. Many people recover recyclable materials and sometimes add value by transforming them into new products. There are almost endless forms of resource recovery that are labour-intense and provide livelihood opportunities.

Pinto and González (2008) demonstrate that operating selective waste collection still costs approximately twice to three times as much as landfilling household waste. Nevertheless, one ton of household waste placed into a triage centre injects roughly 20 R$ (12.6 US$)[1] into the local economy and generates 2 R$ (1.26 US$) in tax benefits (Pinto & Gonzáles 2008). Selective waste collection generates multiple employment opportunities. Taking the example of the Brazilian city Londrina, recycling creates at least 1 direct work post for 1000 inhabitants considering the collection, separation and commercialization of the recyclables. Here the recyclers earn approximately 2 Minimum Salaries (650 US$), which is more than organized recyclers make in most cities in Brazil. In addition, numerous jobs are created indirectly with the recycling industry and sometimes with adding value to specific

[1] All exchange rates are based on the Daily Currency Converter (21.06.2011) of the Bank of Canada.

recyclable products (e.g. transforming plastic PET bottles into washing line - experience *CoopCent* in Diadema) or creating artisanal products from recyclable materials. The following Table 2 provides an example of direct employment through recycling industries in some cities in the metropolitan region of São Paulo. The numbers do not include indirect employment generated through recycling, nor informal business and intermediary activities.

An important concern related to WfE is that with incineration technology, profits don't stay local. As discussed earlier, multinational and large-scale enterprises involved in the technology make the profits, which are mostly transferred into the large centres within the country or abroad.

Municipality	Recycling establishments	Employees	Total population
Diadema	7	76	386,039
Mauá	14	289	417,281
Santo André	5	45	673,914
São Bernardo do Campo	8	360	765,203
São Caetano do Sul	2	13	149,571
Total	36	783	2,392,008

Table 2. Recycling establishments and number of employees. Source: Classificação Nacional de Atividades Econômicas (CNAE), Personal communication Municipality of Diadema, May 2010.

2.4 Recent experiences with WfE in Brazil

Many cities in Brazil are facing increased landfill operating costs or are under pressure to close current waste dumps, which are not adequate to the latest legislation changes. Several municipalities in the state of São Paulo are currently in the process of hiring consultant firms to conduct feasibility studies into waste management. The results are often prescribed WfE technology through PPP financing schemes.

The following table below (Table 3) provides some insight into current WfE developments in Brazil. The data does not claim to be complete. The information sheds light on current trends and practices of governments seeking PPPs to fund WfE technology as a waste management option in their municipality or region.

One waste to energy model currently under discussion in Brazil runs under the bizarre name of '*Tyrannosaurus*'. Inspired by the pre-historic carnivorous dinosaur this facility is meant to triturate solid waste and then generate fuel. The name hints the voraciousness of the process. In addition, the layout of this animal body suggests the various stages of the facility from receiving the solid waste (through the tail), separating the materials and triturating them (in the trunk of the body) to, finally, processing the fuel (in the head of the animal).

The '*Tyrannosaurus*' was acquired in the metropolitan region of Campinas, in the interior of the state São Paulo, from a Finish firm for the cost of 33 million R$ (almost 21 million US$). The facility is meant to burn about 1,000 tons of solid waste per day, generating 500 tons of fuel (Granato, 2011).

City	State	Stage of operation	Proposal
Brasilia	DF		PPP
Campo Grande	MT Sul	Bidding process initiated (July 2010)	PPP (960,000 R$) (604,295.-US$)
Unai	MG	Operating under the name: ("Clean Nature Project").	Solid waste to fuel for state iron smelter and other chemical industries.
Belo Horizonte	MG	Viability study conducted by Pöyry (Nov. 2010).	PPP
Cabo de Santo Agostinho	PE	Thermo-electric plant (to be located in *Mata Atlântica* protected watershed (Pirapam- Tejipió rivers).	PPP (300,000 R$) (188,842.-US$). Concession 20 years. Capacity: 2.856 tons/day to generate 27MW/day.
Rio de Janeiro	RJ	Pilot plant called 'Usina Verde' (Green plant) already operating at the Federal University of RJ, incinerating 30tons/day.	Ashes are used in floor tile production and could be used in agriculture to reduce soil acidity.
Barueri	SP	Bidding process opened on: 20.10.2010	PPP (Concession 30 ys.). Planned capacity of 750 tons. To be installed at the former landfill.
São Sebastião	SP	Viability study conducted.	PPP. 6 firms have placed a bid[2].
São Bernardo do Campo	SP	Bidding process concluded. In process of getting approval.	PPP. (220,000 R$) (138,484.-US$). Proposed location at former waste dump Alvarenga.
Ferraz de Vasconce-los	SP	Consortium of the Upper Tietê river, in collaboration with Suzano city (Nov. 2010).	PPP (200,000 R$) (125,894.- US$). Capacity of 700 tons/day to generate 30MW/day.
Santos	SP	Viability study has been conducted (July 2010).	PPP (300,000 R$) (188,842.- US$).
Campinas	SP	Known under the name '*Tyrannosaurus*'. Operating in Campinas metro area (Sumaré, Hortolândia, Nova Odessa, Americana, Sta. Bárbara d'Oeste, Monte Mor).	PPP (33,000,000R$) (20,772,624.-US$). Solid waste to fuel. Burns about 1,000 tons of solid waste/day, generating 500tons/fuel.

Table 3. Brazilian municipalities with proposed WfE plants (2010). Sources: Unpublished literature searched on the Internet (last accessed 28.03.2011).

[2] WfE: 1.) EMAE 2.) Keppel Seghers, Singapure, 3.) Consortium AEMA / FAIRWAY / SENER, multinational 4.) HERHOF / GPI, Germany, 5.) LIXOLIMPO CONSULTORIA AMBIENTAL, multinational, 6.) DEDINI Indústrias de Base.

3. Selective waste collection and recycling

In cities in Brazil, as in most countries in the global South, a large number of informal recyclers (*catadores)* collect recyclable material from the garbage. Often the recyclers establish partnerships with households or businesses separating the material for regular pick up. There is no exact record about the number of *catadores*, nor about the quantities of material that they recover on a daily basis, since the numbers fluctuate significantly over time.

In many municipalities the recyclers are organized in cooperatives or associations and perform selective waste collection in partnership with the local government. The level and continuity of the official support varies among these experiences, from governments simply tolerating the work of the recyclers to remunerating the collection service performed by the *catadores*. Although governments might have taken important steps to implement inclusive selective waste collection, these programs are often prone to discontinue after election periods. When recycling programs are consolidated within the local community and when public policies protect these cooperative recycling schemes the work is valued and the results are more successful.

In 2010, approximately 8% of all municipalities in Brazil (443) had established a selective household waste collection, which reflects a steady increase since 1994, when only 81 cities had recycling programs. More than 62% of these municipalities collaborate with organized recycling cooperatives in the collection and separation of the materials (Compromisso Empresarial para Reciclagem) [CEMPRE], 2010).

The cost for selective waste collection is still 4 times higher than the regular collection costs. Nevertheless this value has been steadily decreasing, from being 10 times more expensive in 1994. There are also very large discrepancies between different cities, with Londrina having the lowest cost per ton of recovered material (7.2 US$/ton), compared to São Bernardo do Campo (575 US$/ton) or Florianopolis (389 US$/ton), for example. Most selective waste collection programs are located in the southeast (50%) and in the south (36%) of Brazil. The amount of material recovered through official selective waste collection programs varies a lot between each city, with over 3,500 tons/month of recovered materials Londrina is taking the lead, followed by Porto Alegre (2400 tons/month), Curitiba (2228 tons/month) and Brasilia (1327 tons/month) (CEMPRE, 2010).

According to the data promoted by CEMPRE (2010), the average material composition of selective waste collection in Brazil contains approximately 13.3% of unrecyclable materials. The rest is composed of 39.9% paper and cardboard, 19.5% plastics, 11.9% glass, 5.7% other recyclable materials, 1.9% tetrapack (combined plastic, aluminium foil and cardboard), 0.9% aluminium and 0.2% electronics. Most of the plastic, 36.2% comes as mixed material, 27.1% as PET, 16.9% as PEAD, 9.7% as PP, 6.3 % as PVC among other plastics (CEMPRE, 2010).

There are very little experiences targeting formal collection of organic household waste. A pilot study on door-to-door collection of compostable, organic household waste was conducted in the city of Diadema, confirming the potential to generate income and produce rich compost for urban agriculture and gardening activities (Yates & Gutberlet, 2011a, 2011b). Experiences from Cuba and Argentina underline the possibilities of contributing to food security by collecting and composting clean organic waste from the households (Mougeot, 2005). Further empirical studies are needed to advance this particular form of resource recovery, as important contribution towards *zero waste*, the avoidance of any waste generation. The following two examples showcase the possibilities in terms of inclusive waste management, generating income and recovering valuable resources.

3.1 The case of Londrina

Since 2001, Londrina's *Reciclando Vidas (Recycling Lifes)* program has become a benchmark for selective waste collection in Brazil (Suzuki Lima, 2007). Londrina is located in the state of Paraná, in the south of Brazil. The city services 90% of its almost 500,000 inhabitants, with an adherence rate of 75% of the population. In 2010 these numbers translate into 26.6% of the household waste being recovered through selective collection, separation and recycling. Only 4% of the material collected in this program is considered unrecyclable, which is particularly low when compared to other municipalities who are struggling with up to 50% of rejected material. Door-to-door collection allows for a direct contact with the population, a key aspect in improving the quality of the selective waste collection. In Londrina continuous community environmental education performed by the recyclers has reduced the percentage of rejected material from 15% in 2001 to 4% in 2005. In addition, here the recyclers work on tables and not on assembly line belts to do the classification, which also contributes to the reduced loss of materials. A study has shown that separating on tables generates approximately 5% rejected materials, whereas the assembly belt produces between 25 and 30% rejected materials (Pinto & González, 2008).

Today approximately 500 *catadores* work in this local resource recovery program. The city is divided into 33 sectors and has 33 triage centres. In 2011 the recyclers were paid 64.00 R$ (40.29US$) per ton of commercialized, recycled material by the government for the quantity of material collected. In addition the recyclers receive a monthly amount of 33,000.00 R$ (20,772.-US$) for the service of selective collection, prolonging the life of the landfill. This value is divided amongst the recyclers according to their work effort. In 2010, the municipality has in addition invested approximately 20,000.00 R$ (12,589.-US$) every month to acquire trucks and electric selective collection carts improving transportation.

Recent numbers demonstrate a steady expansion of the program from 156,927 kg/month collected from 60,000 households in March 2010, to 274,411 kg/month from 71,648 households in July 2010. The material is separated into 25 different categories and sold to the industry. Cardboard, newspapers and other papers make up the largest quantities of the materials collected, followed by broken glass, PET bottles, tetrapak, and thin coloured plastics.

As part of the *Sustainable Waste Management Project* (PSWM) delegates from the local government and recyclers visited this experience in selective waste collection in Londrina, Paraná. Participants highlighted the existence of a:

- complex and fair payment system of the recyclers performing various tasks in recycling,
- high level of commitment of the local government with the selective collection system,
- consolidation of public policy more than just a government program,
- high level of feasibility of the door-to-door collection system,
- contractual relationship between government and recyclers,
- strong communication system in the community (e.g.: the recycling program offers a *800 number to communicate with the population),
- exemplary transparency and trust between collectors and government,
- high self-esteem of recyclers.

As possible difficulties the participants identified:

- precariousness in the infrastructure of the triage centres,
- problems in the logistics of integrating with the recycling industry,
- recyclers seem not yet to be fully inserted into the recycling process,
- current shortcomings in the program for occupational health and environmental education.

The experience of Londrina showcases the importance of collaborative policy programs between government, recyclers and community. Good stakeholder integration is crucial for achieving success. Moreover, adequate public policy for integrated waste management must be in place, more than just a selective waste collection program, but continuously and beyond four-year government periods.

3.2 The case of Diadema

The city of Diadema is located in the Greater Metropolitan Region of São Paulo, with 370,184 residents in 2010 (Instituto Brasileiro de Geografia e Estatística [IBGE], 2010), covering an area of 30.84 km^2. Diadema has the third-highest population density in Brazil, with over 11,000 inhabitants per square kilometre. It is primarily a low to middle-class, industrial city and approximately 25% of the population is housed in favelas (squatter settlements), which occupy 3.5% of the municipality. Since only 73, 225 people out of the total population of Diadema are economically active in the formal labour market, many residents are threatened by poverty, food insecurity and unemployment. However, Diadema represents a progressive political scenario in Brazil, providing opportunities for participation and political change.

In 2004, the local government initiated the *Vida Limpa* (Clean Life) program, a city-wide recycling programme that is operated by recycling associations (Gutberlet, 2008a, 2008b). In 2008 there were six fully-functional collection depots established across the city, based on catchment areas. The recyclers organized under the umbrella association of *Pacto Ambiental* (Environmental Pact). In June 2004, Diadema became the first municipality in the country to support recyclers' associations with an official policy of remuneration. As of 2008, the *catadores* received 38 R$ (24 US$) per ton of material diverted from the landfill, under a municipal partnership memorandum. Remuneration contributes to the average income of 380 R$ (approximately 239 US$) per month amongst *Vida Limpa catadores* (Gutberlet, 2008b). Despite the pro-active policy in place, recyclers in Diadema remain physically and socio-economically vulnerable, dependant on unstable economic markets.

In 2007 the city generated 7,514 tons of household waste every month, of which 36% (2,705 tons) are inorganic recyclables. In 2007, the program collected more than 44 tons every month. The evaluation of the door to door selective collection, conducted in 2007, highlighted the fact that in some of the neighbourhoods the recyclers only collected the material from those houses contacted initially, which left out a significant part of the households. Furthermore, one of the neighbourhoods, *Chico Mendes*, had an extremely high level of rejected material (51%), as a result of low awareness among the population of the program. There was little interaction of the recyclers with the locals to raise the awareness. Most of the other neighbourhoods had a rejection level between 1% and 11% (Gutberlet & Takahashi, 2007). Since then the program has undergone growth and retraction, depending on the level of support received by the local government. Since 2011 the *Vida Limpa* program also recycles discarded wood and cooking oil and the municipality is currently working on the expansion of a pilot project on organic waste for composting and community gardening (Yates & Gutberlet, 2011a, 2011b).

4. The new Brazilian federal solid waste legislation

At the end of 2010, a new federal legislation on waste management (Law #12.305/2010) was sanctioned. This recent law and the specific legislation piece on basic sanitation (Law # 11.445/07) establish the overall rules and obligations for solid waste management in the country. In addition, more specific laws, such as the resolution CONAMA (#316/2002) and specific state legislation deal with the establishment of sound solid waste facilities.

Over the past 11 years the organized recyclers, through the inter-ministerial panel working on solid waste management, have actively been involved in the design of this legislation. There are obviously also other stakeholders involved in the process who defend different interests and are able to manifest higher levels of power.

This law is innovative in institutionalizing selective waste collection, in recognizing the recyclers (*catadores*) as important social actors in solid waste management, and in promoting shared responsibility, through sector accords and reverse logistics. The law requires every municipality to develop their own solid waste management plan, besides plans on the regional and inter-municipal levels. The law foresees specific funding possibilities for municipalities to be able to access the resources required to upgrade their waste management plants. The law further reiterates the hierarchy in waste treatment from not generating, reducing, reusing disposed products, to recycling and composting and final disposal of waste at landfills.

Several challenges are given with the new legislation, including monitoring its application and adjusting the legislation to the reality of the *catadores*. This means to guarantee the recognition of recyclers as professional category and insert the *catadores* in the reverse logistics. The Government will have to be the facilitator in the dialogue between recyclers and industries.

The participation of organized recyclers is mentioned various times throughout the legislation. Article 40, highlights the need to prioritize contracts with recycling coops or associations in the selective collection of waste and in the implementation of reverse logistics. Article 41 and Article 42, guarantee the contemplation of organized recyclers in the city's waste management plan and reassure addressing the needs for these groups to participate in the implementation of the programs and actions defined under that plan. The federal government needs to regulate specific programs to improve the working conditions of the recyclers and to generate opportunities for their social and economic inclusion, according to Article 43. Recyclers' organizations are exempt of the bidding process (Article 44).

The initiatives highlighted under Article 42 will be benefitted by:

- fiscal, financial and credit incentives;
- availability of public land;
- directing the solid waste from federal public institutions to recycling coops, as defined under the Decrete # 5940/2006[3];

[3] Presidência da República. Casa Civil Subchefia para Assuntos Jurídicos, Decreto No. 5940, 25.09.2006, Available from http://www.planalto.gov.br/ccivil_03/_Ato20042006/2006/Decreto/D5940.htm

- economic subsidy;
- setting criteria, aims and other complementary forms to seek environmental sustainability during the implementation of public contracts;
- payment for environmental services;
- support in the definition of clean development mechanism projects or any other mechanism based on the UN climate convention.

Furthermore, Article 81 highlights the fact that federal funding institutions may create specific funding lines to support coop recycling, by facilitating the acquisition of equipment used in solid waste management.

Nevertheless, there are also some serious flaws to the law. Shortly before sanctioning the law a significant alteration in the wording of Article 9, Paragraph 1 was made, allowing for incineration as WfE to play a more prominent role. Originally the text read: *"Technologies that recover energy from waste can be used if their technical and environmental viability has been proven and with a system in place to monitor gas emissions, approved by the environmental agency"*. Originally the sentence continued with the following wording, restricting the use of incineration *…"after all other waste management possibilities, mentioned earlier have been exhausted"* (Art. 9 § 1º) (Translated by the author). By taking away this part of the sentence a change in hierarchy now allows for incineration as an energy recovery measure, even before reuse, composting and recycling has been performed.

Furthermore, under Article 58 the law excuses governments from including *catadores* in waste management, when the existing recycling organization is inefficient or for some reason presents public health hazards. Unfortunately most of the recycling coops and associations still perform their work under precarious and often unhealthy conditions. Many recycling groups, particularly those that are not supported by local governments are still extremely vulnerable, with inefficient infrastructure, weak administration, lack of organization, and low remuneration. With little or no backing from the government, their operations are often not viable, generating very low income (sometimes half a minimum salary[4]) and as a consequence there is a high fluctuation rate of participant recyclers. If by any means the government declares that the existing coop does not have the capacity or are considered economically inefficient to participate in the selective waste collection, then the groups can be excluded.

Another important social shortcoming of the new legislation is the fact that autonomous recyclers are excluded as participants in resource recovery. Similarly there is also no mentioning of the role of scrap dealers who usually buy the material recovered from the autonomous recyclers. With the new law in place informal recyclers will be twice excluded, since most of them already barely survive from the payment of the recyclable materials. There is no reference as to how these *catadores* could be included in waste management in the future.

5. Conclusion

Not only does incineration and energy recovery from waste cause environmental hazards, but it also dismisses the fact that resource recovery and recycling schemes are more socially and environmentally friendly than simply burning the materials. Recycling generates a large

[4] One Minimum Salary is currently 540.- R$ or 325.- US$ (March 30th 2011).

amount of employment and contributes to resource conservation. Recovering recyclable materials generates net carbon credits, which should be redirected to those who are engaged in the collection, separation and recycling of waste. Recycling could also tap into *'Carbon gaining funds'* because that is what reuse and recycling does. Such a measure would be more socially and environmentally just and, at the same time, it would also contribute to addressing poverty reduction, one of the Millennium Development Goals.

The prevailing mainstream discourse and general dominant politics, however, still support the growth oriented economic development model, understanding progress as unlimited growth. This dominant view perceives resources as limitless and defends the rational that there will always be unrecyclable waste as a result of production and consumption. Consequently, the large quantity of rejected material and the growing demand for energy are seen as justification for the implementation of incinerators as energy generating plants.

As an alternative path, however, participatory sustainable waste management (PSWM) translates into the networking among different stakeholders and the construction and strengthening of solid waste management policies with the inclusion of the recyclers, aiming at social equity and environmental sustainability. This form of management is based on the principles of social economy, valuing and empowering the recyclers, aiming at reducing, reusing and recycling; addressing responsible consumption and refusing the waste of resources. Few examples demonstrate that this path can be viable, despite many difficulties still have to be overcome. Cities like Diadema and Londrina are experimenting with PSWM and have achieved important results towards resource recovery.

There is an eminent need to address the challenge of reducing waste generation. It is a given that most municipalities in the global North and also now in the global South are threatened by a shortage of landfill capacity. It is also common sense that the fact that environmental damage can be caused by inappropriate waste management needs to be addressed. In addition, concerns regarding global warming and resource depletion related to production and consumption are looming at the horizon. It is essential to adopt a cyclical materials flow approach. *Waste for energy* is not a cyclical process, since the material dies with incineration. Participatory sustainable waste management facilitates the cyclical use of resources, generates work and employment and cares for the environment and future generations. Further insights are needed to explore the potential of resource recovery to promote a paradigm shift towards *zero waste* and the formation of more sustainable societies.

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Part 3

Industrial Solid Waste

13

Solid Waste Utilization in Foundries and Metallurgical Plants

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1. Introduction

The issue of waste management and utilization in foundries and metallurgical plants covers a lot of completely different materials in various forms (solid, liquid or gaseous). In this chapter, solid waste utilization is described based on the experiments and industrial experiences of Department of Foundry, Silesian University of Technology. The first part of the chapter introduces the readers into the subject of pneumatic powder injection into liquid metal process. It is a method widely used to utilize solid wastes in foundries and steel plants giving good technological and economic results. The casting production process is inseparably connected with pollutants emission into the environment, that is into air, water, soil and also noise emission. The fumes and gases from coke-fired furnaces are deposited in the air as well as other pollutants created when metal is molten in electric furnaces. Their amount can be limited by use of the modern high efficiency filters but the amount of dusts to deposit on waste dumps consequently increases. The water contaminations are caused by open melting furnaces cooling systems. The solid wastes from various production stages (moulding mass de-dusting, furnace de-dusting, blast cleaners de-dusting, slag etc.) are deposited on waste dumps. The latter can be utilized after granulation process as a road building material whereas furnace dusts are treated in recirculation into furnace systems decreasing their final quantity and improving utilization of some important elements, mainly iron (Fiore et al., 2008; Lee & Song, 2007; Salihoglu et al., 2007; Fu & Zhang, 2008).

2. Powder pneumatic injection into liquid metal

Materials introduction into foundry furnaces where there is a solid charge at the beginning and liquid alloy at the end of the melting process, can be operated by many ways. The introduction method depends on furnace construction (cupola, electric induction furnace, electric arc furnace etc.), the form of the powder introduced (dust, granulate, briquettes) and its chemical composition and foundry plant mechanization level (Holtzer et al., 2006; Jezierski & Janerka, 2008).
The most often used are:

- introduction by hand for the small furnaces and small quantities of materials introduced (chemical composition correction),
- mechanical introduction with use of vibratory conveyors into charging hopper or dosing devices. Most often blocks or briquettes are introduced this way along with the solid charge,

- pneumatic introduction of powdered material with carrier gas. This method is one of the pneumatic conveying applications. The liquid metal inside furnace or ladle replaces the typical pneumatic conveying receiving device. Powdered material is directly introduced into metal bath by means of pneumatic feeder and through pipes ended with an injection lance (Holtzer, 2005).

The two first methods mentioned require a special material pre-treatment which means it must be de-dusted or granulated or briquetted. They are not appropriate for the introduction of dusty fractions because of possibility of environmental pollution and the inefficiency of the metallurgical processes (the dusts are easily sucked out of furnace). Moreover, it should be emphasized that the most commonly used are waste materials in form of dust and their granulation or briquetting requires additional devices which increases total production costs. During pneumatic injection, the use of fine material particles causes a large contact surface between them and liquid metal and consequently high dissolution rates of material. Additionally, the lance is introduced inside liquid bath that eliminates environment dustiness problem and process is intensified by particles and metal mutual movement, forced by carrier gas stream. Finally, it causes significant physical chemical processes rate increase when compared to hand operated or mechanical introduction. These advantages caused that powdered materials pneumatic injection is used in the following processes (Jezierski & Janerka, 2001; Cholewa, 2008):

- liquid cast iron recarburization inside electric arc furnaces and cupolas (the solid wastes from carbon materials production processes can be utilized),
- desulphurization and dephosphorization of alloys inside ladles and electric arc furnaces,
- alloy additions introduction into liquid metal inside ladles, electric arc furnaces and cupolas (the possibility of utilization of dusts from alloy additions production),
- alloys inoculation or refining and liquid composites production,
- inoculants introduction inside the liquid metal stream during mouldings pouring in,
- slag foaming inside electric arc furnaces during steel production,
- dusts recycling from cupolas and electric arc furnaces de-dusting systems,
- coal dust injection into blast furnaces,
- waste plastics utilization by their injection into blast furnaces.

Many factors determine the correct powder pneumatic injection process. These are: pneumatic transportation parameters, carrier gas and material mass flow, solid-gas mixture mass concentration, gas and particles velocity on the lance outlet. These parameters depend on feeding device construction which should give a possibility of changing individual parameters. The liquid metal parameters (the initial bath temperature, chemical composition, metal bath mass), the grade of powdered material and carrier gas play together an important role, too (Engh & Larsen, 1979; Janerka & Jezierski, 2002). The kind of carrier gas used depends on the process itself, the reagent being introduced and the furnace. The powdered material carriers are usually: compressed air, argon or nitrogen. When carbon materials or dusts are introduced air is mostly employed. Inoculants introduction, desulphurization and alloy additions introduction into ladle requires argon usage. When compressed air is used (because of its dampness) the filters, dehydrators or driers are used. The powdered materials introduced into liquid metals can be divided into: powders insoluble in liquid metal (forming slag) and soluble reagents which are assimilated by metal or refine it. This is both for materials utilized earlier in metallurgical processes and for dusts recycled from various production processes. Powders are characterized by physical chemical properties as melting point, gas saturation and solubility inside liquid metal. Their dampness should be minimal (<0.1%) because of possibility of hydrogen assimilation by liquid metal. In order to design devices properly and select pneumatic injection parameters properly, the bulk density and compactibility (the level of density) of the injected materials must be known. It is important to ensure that the material will not suspend in feeders and silos which can cause instability in dosing devices. This is particularly important for dusts created in metallurgical furnaces which possess very strong internal bonds (Janerka, 2010; Kanafek et al., 1999). As mentioned earlier, the important element of the powder injection process is a feeder, where the mixing of carrier gas and powder as well as subsequent diphase stream conveying take place. The powder injection setups used nowadays are of various constructional and functional designs. The powder feeders should be characteristic for powder feeding stability, small carrier gas consumption and be hermetic. The feeders can be divided into two groups – gravitational and pressurized. The gravitational ones work on loose powder pouring basis. The material portioned with mechanical feeders (with sectors, cells or feeding screw) is introduced into pipeline and transported with carrier gas stream. Because the feeders are not completely hermetic when the overpressure on lance outlet appears (metallostatic pressure), these feeders can be used only when the powder is introduced solely on the liquid metal surface. In pressure feeders the material into pipeline introduction is intensified by overpressure in the upper part of the feeder. Solutions like that are used when the material is transported through longer pipeline and when the injection lance is submerged into liquid metal (Janerka, 2010; Kokoszka et al., 1999). One of the features that differ the classic pneumatic conveying (where the receiver is a silo) from pneumatic injection process (where the receiver is liquid metal) are the working parameters. For the pneumatic injection the range of transportation parameters differs from those for pneumatic conveying. The pneumatic conveying is determined mostly by the economic factors which can be obtained for high solid-gas mixture mass concentration (this parameter is a quotient of material mass flow and gas mass flow) and small diphase stream velocity. In injection process technological parameters play the most important role. Therefore high flow velocity on the lance outlet (Approx. 70÷120 m/s) to ensure high stream energy to achieve its deeper penetration into liquid metal is applied. The mixture mass concentration is usually from 8 to 20 kg of solid per kg of gas (Janerka, 2003).

2.1 Recarburization of liquid metal

The projects targeted to limit waste generation may cover the solid metal charge for cast iron production change and the pig iron can be replaced with steel scrap. The pig iron share may reach up to 75% of the charge mass. The pig iron is usually blast furnace product and the carbon content varies from 3.5 to 4.5%C. There are two main pig iron grades: steelmaking pig iron and foundry pig iron which are supplied to foundry in the form of pigs. The foundry pig iron can be further divided into: hematite, semi-hematite, normal, phosphoric and special for ductile iron production. The pig iron production process generates significant waste amounts and is energy-consuming. It is possible to produce cast iron with no pig iron at all (synthetic cast iron). In this case it is necessary to compensate carbon deficit when the pig iron is replaced by steel scrap. The greater utilization of the steel scrap which is waste material may be considered both from the ecological and the economic point of view. The price of steel scrap depending on the world situation is three times less than foundry pig iron. The cast iron smelting on the steel scrap base only forces to more than 3% carbon content increase. In this case the carburizer mass should be around 3.8 to 5% of the metallic charge mass. This amount depends on carburizer grade and the recarburization method employed. On the basis of the estimated carbon content in grey iron, steel scrap, pig iron and carburizer one can proceed with the calculations of the specific charge materials. To reach 3.2%C content in iron when the heat is made only with the steel scrap, 74% pig iron and 26% steel scrap should be charged. When 100% of steel scrap is used and the goal is 3.2%C in the final alloy, the introduction of 4.2% carburizer is necessary. These proportions may vary of course when some portion of the process scrap with the carbon content Approx. 3.2% is introduced into solid charge (Skoczkowski, 1998; Janerka, 2010). The most often used carburizers are natural graphite, anthracite, synthetic graphite and petroleum coke. Graphite is a natural mineral and occurs as a 72-80% of carbon rich ore. Its natural colour is glossy black or steel black. Dependably on amount and kind of impurities in ore the natural graphite is produced by means of special enrichment. It may be achieved by sorting inside the air stream and flotation (Janerka et al., 2009). Anthracite is a product of high plant substances carbonification which contain of 92-97% of elemental carbon. It is characterized by tar black lustre, high mechanical strength and low volatile parts content of 3-8%. Synthetic graphite is the name given to graphite obtained during high-temperature process (graphitization) of the coke (petroleum, coal or pitch) and anthracite. The properties of the synthetic graphite and its structure degree of order depend on both input material and the final treatment temperature. Petroleum coke is a solid carbonaceous product obtained during thermal treatment of the oil distillation residues. The input product for the coking are heavy residues from various stages and methods of the crude oil refining (Janerka et al. 2009; Janerka, 2010).

The production of those materials is in some degree connected with environment pollution. It should be emphasized that the necessary amount of carburizer to produce 1t of cast iron is relatively small and equals 40-50kg. The most environmental friendly carburizers are natural graphite and anthracite. These are minerals which are only mechanically ground and calcined when only the volatile parts and some sulphur compounds are emitted into the atmosphere. Petroleum coke and synthetic graphite, which require high-temperature processing are much more nuisance to the environment. Materials from the scrap graphite or carbon electrodes grinding process, materials from graphite linings used in various industry branches offered by many suppliers, can also be used as carburizers. Foundries which possess electric arc furnaces we can use their own scrap electrodes as a carburizer . The production wastes from electrode manufacturers can be also used as carburizers. Mostly these are dusts from electro-filters with very big carbon content of 97-99%C.

The essential parameter which characterizes the recarburization process from the technological point of view is its efficiency (recarburization effectiveness, carbon by liquid metal assimilation ratio). This parameter determines recarburization time and the carburizer amount to be introduced to obtain the planned carbon content increase. The efficiency (the effectiveness) of the recarburization is given by equation (Janerka 2010, Chojecki et al. 2002):

$$E = M_m \frac{C_k - C_p}{M_n \cdot C_n} \cdot 100\%. \tag{1}$$

where: C_p – the initial carbon content, %, C_k – the final carbon content, %, M_m – mass of metal, kg, M_n – mass of carburizer, kg, C_n – carbon content in carburizer,%.

The carburizer introduction can be realized by its addition into solid charge, onto liquid metal surface, onto liquid metal stream or on the ladle bottom. In these cases the carburizer granulation should be something between 1 and 6mm (with no dust in it). The carburizer introduction into solid charge can be realized in electric induction and arc furnaces and cupolas, too. It is a method which does not require any additional investments to buy the recarburization devices. The high recarburization level can be achieved by this method with no melting time extension at all. The use of that method allows not only to correct the carbon content in alloy but to produce the synthetic iron, too. Therefore it is often used for the cast iron production.

The carburizer introduction onto liquid metal bath surface is the most common recarburization method for the electric induction furnaces both for synthetic cast iron and for cast iron made on a pig iron production method. It is because after the solid charge is melted the sample for chemical analysis is taken and on its basis the real carbon deficit is estimated. Moreover, in the electric induction furnaces when the charge is molten, the continuous electromagnetic stirring occurs what causes an increase of the process efficiency.

For the cupolas and electric arc furnaces the carburizer introduction in carrier gas stream is often used and the fine (dusty) fractions of carburizers can be utilized.

The recarburization setup example is presented in Fig. 1. Its main part is a pressure container (1) of 0.25 to 1.0m^3 capacity (Janerka, 2010; Kanafek et al., 1999; Kokoszka et al., 1999). The bell door is situated in the upper part, whereas the mixing chamber (3) is situated at the bottom. The pressure container is equipped with decompression valve which allows to decompress the container when the working cycle is finished. The air pressure above the material being conveyed is regulated by means of the reducer (4). The air supply is started and stopped with the main valve (10) operation. All the valves can be operated from container control switchboard (2) or from the control panel situated in the furnace control room. The container is mounted on the extensometric scales (5) and their recordings are displayed on the control switchboard (2). The carburizer is transported through a pipe (11) ended with the injection lance (13) inserted into electric arc furnace (14) and submerged into liquid metal. The lance can be mounted on the manipulator (12) which allows its automatic introduction into liquid metal. This method makes the work safer and guarantees better process repeatability.

The silo (7) with the carburizer can be situated above the feeder and it can have 24hours or shift working capacity. The screening sieve (8) should be mounted at the top of container to stop the impurities and oversize. The bottom silo (7) part is a pneumatically driven damper (slide or swivel). Between damper (6) and chamber feeder (1) the compensator is necessary to eliminate silo influence on the weighing system. The carburizer is supplied by manufacturers mostly in big-bags of 1m^3 capacity. On the basis of the author's experiments (Janerka 2010; Janerka et al. 2010) and literature overview (Kosowski, 1982; Przeworski, 1986) an analysis of the influence of the introduction method (SC – carburizer into solid charge addition, S – addition onto metal surface, PI – pneumatic injection), furnace type (IH – induction furnace, EAF – electric arc furnace) and carburizer grade (GS – synthetic graphite, GE – ground electrodes scrap) on the recarburization efficiency was carried out (Fig. 2.). Of course these are average results, because there are many factors which can change those values all the time. However, some conclusions can be put forward. When the recarburization efficiencies recorded in induction furnaces for the synthetic graphite and ground electrodes scrap are compared, it can be seen that they are even higher for the electrodes scrap.

It is probably an effect of the recarburization method employed but it show that this waste material makes up the full value of carburizer. The next remark is connected to the recarburization method. The introduction of carburizer with solid charge in induction furnace allows the foundry to achieve the efficiency respectively 92 and 83% and for the addition onto metal surface the process efficiency is reduced by Approx. 6%. For the EAF when the carburizer is added with solid charge the efficiency is 10 to 15% less than for induction furnace.

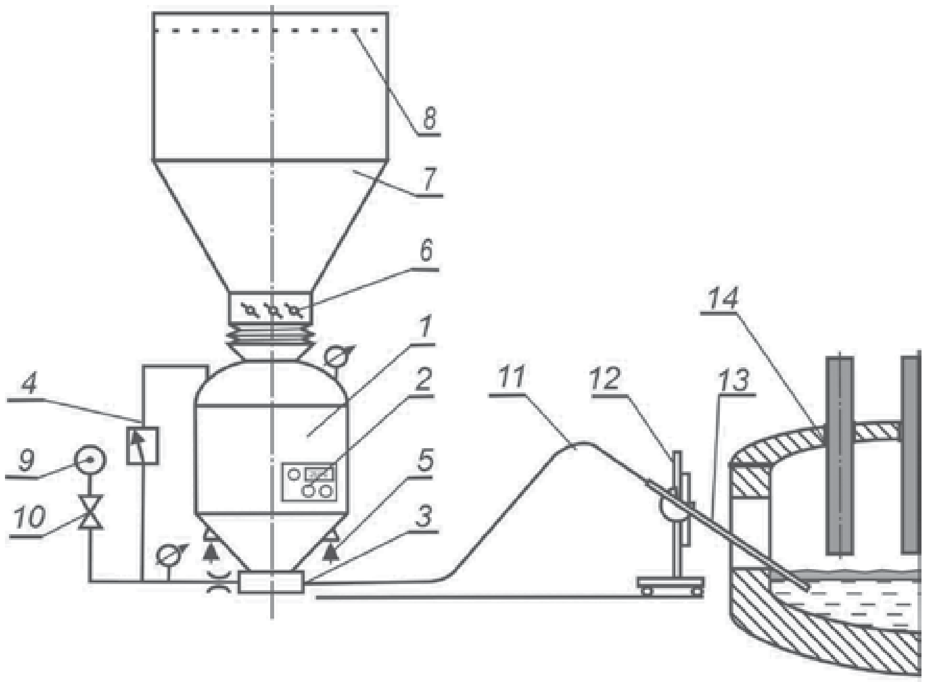

Fig. 1. The recarburization setup for the electric arc furnace: 1-pressure container, 2-control switchboard, 3-mixing chamber, 4-reducer, 5-extensometric scales, 6-slide damper, 7-silo, 8-screening sieve, 9-the compressed air supply, 10-main valve, 11-pipe, 12-lance manipulator, 13-injection lance, 14-electric arc furnace

For the surface carburizer addition in EAF the recarburization efficiency is at most around 53%. It should be mentioned that such a result can be achieved only after tens of minutes because of very slow liquid metal movement inside furnace. The process can be accelerated by mechanical stirring but it is hard to do so. Very high efficiency level in EAF can be achieve with the use of pneumatic powdered carburizer injection. The 80% efficiency is recorded just after few minutes after the material has been completely introduced. Our researches have shown that the diphase stream parameters have strong influence on the efficiency and rate of the process (Janerka, 2010). These parameters depend on the feeders construction. Nowadays the devices allow the control of mass gas flow in the range from 0.03 to 0.20kg/s. This parameter directly influences (when the geometry setup does not change) on the solid-gas velocity on the lance outlet and consequently on the stream dynamics. The particle velocity inside the pipe can be calculated as a product of the air velocity and carburizer particle slip coefficient in relation to carrier gas. This coefficient s is in range s=0.5 to 0.8. Low gas flow causes low velocity and stream energy onto lance outlet and as a result the limited reaction zone between carburizer and liquid metal (small stream surface). Large carrier gas flow ensures good stream dynamics but simultaneously as a cooling effect significant liquid metal temperature decreases and more intense carbon oxidation occur (when the carrier gas is compressed air, larger oxygen amount is introduced). Too large carrier gas flow can cause a carburizer dozing decrease in some device designs. The material mass flow can be changed in range m_c = 0.10 to 2.0 kg/s. Small device output increases recarburization efficiency but extends injection time what causes liquid metal temperature decrease. Large mass flow causes some carburizer portion not to be assimilated by liquid metal and floats into surface. It is an important recarburization process index, too. Of course, these two parameters mainly depend on furnace capacity which is directly related to the conveying pipe diameter. Badly chosen gas and material mass flow cause recarburization decrease to 40 to 50%. Undoubtedly pneumatic recarburization in the electric arc furnaces is the only method to achieve high indexes of that process.

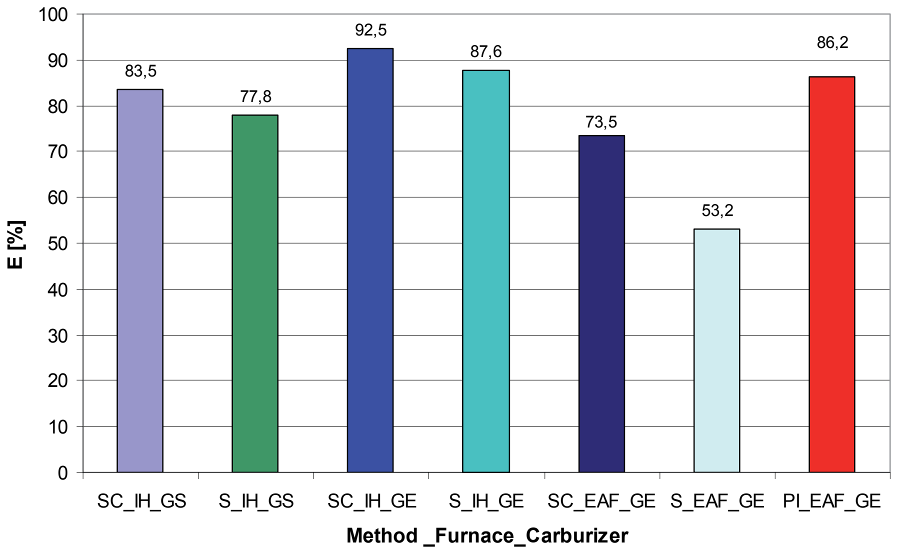

Fig. 2. The influence of recarburization method, furnace type and carburizer grade on the recarburization efficiency

2.2 Powder and dust injection into cupola

According to literature data (Ratkovic & Dopp, 2004; Smyksy & Holtzer, 2002, 2007) and authors' own experiences the cupola melting process creates dust in amount of between 4 kg/t and 15 kg/t of molten cast iron depending on the charging materials, furnace type and mass of cupola coke used (or not for coke-less cupolas). In Germany alone cupolas generate over 30000 t of dust per year. The dust being sucked out includes many valuable elements which are additionally very harmful (Zn, Pb, Cd). The Fe content is usually higher than 10%, so the dust itself is a valuable charging material. When the dust contains > 15%C it can be an extra fuel, too.

Since, nowadays a bigger and bigger part of the charge materials for cupolas (sometimes up to 40%) comprises automotive scrap, mainly zinc coated sheets, the high Zn content in cupola dust appears a serious problem. The zinc content in the dust may achieve up to 20% what means it can be considered as a charge material in zinc metallurgical plants. Moreover, repeated recirculation of dust into the cupola causes an increase in the economic factors of the process.

At the Department of Foundry experiments of re-injection of cupola dust together with the finest fractions of ferrosilicon and anthracite (considered as wastes) were carried out and resulted in several industrial installations. The experiments were conducted with use of experimental setup built in the Department of Foundry and one of the most important results was industrial installation implemented in cast iron foundry, see Fig. 3 where its scheme was presented.

There are generally two methods of cupola dust treatment (when it is going to be re-used in furnace), the first is direct pneumatic injection back into furnace and the second is its briquetting and introduction into furnace in this form (Smyksy & Holtzer, 2002). The Department of Foundry of Silesian University of Technology has been involved in the experiments of pneumatic dusts re-injection into cupolas for several years. Their effect, again with cooperation with POLKO company was designing and implementing of several installations that are described in the paper on the example of Czech foundry based on cupolas.

The mentioned foundry wanted to solve the problem of simultaneous injection of three materials of different characteristics:

- cupola dust,
- small fractions of FeSi (considered as wastes),
- pulverized anthracite.

Therefore the first step of experiments was developing the best mixture recipe both from metallurgical process (final carbon content in cast iron, impurities level etc.) and technological (pneumatic conveying parameters, estimated temperature drop etc.) point of view. Then the preliminary tests were carried out when the materials listed above and their mixtures: anthracite + FeSi (50% + 50% mass) and cupola dust + FeSi (50% + 50% mass) were used. The pneumatic injection installation was based on pneumatic chamber feeder of V_n = 0.25m^3 capacity, see (Kanafek et al., 1999). The pneumatic chamber feeder was equipped with electronic control system and a precise dosing system within the required flow range (2÷5 kg/min). The feeder's mass changes (during injection process) were continuously recorded with ±0.1kg accuracy which enabled to quickly estimate the powdered material outflow and in the same way the efficiency of the injection installation in the real time manner. Apart from the feeder, the installation consists of the elastic pipe of L=25m length and d_w=0,025m inside diameter from pneumatic feeder to the end of installation (injection lances integrated with cupola nozzles/tuyeres). Moreover, some important constructional changes in the mixing chamber (situated at the bottom part of the feeder, where the powdered material mixes with the carrier gas) were made. The porous liner to fluidize of loose material inside the container was situated at the bottom part of pneumatic feeder.

From the technological point of view not only pneumatic conveying parameters but the transportation stability during the injection cycle was crucial. After some design changes and parameters adjustment both results were achieved and for the powdered material mass flow m_c = 2÷5 kg/min (well inside the requirements) the working cycle remained stable.

The implemented injection system integrated with cupola nozzles made utilization of the whole mass of dust from dust extraction system possible and the injection process did not negatively affect the produced alloy quality.

The implementation of the waste utilization method based on pneumatic injection in the aforementioned foundry enabled the recovery of waste materials received during

metallurgical processes. Earlier this material was exported from the plant to special utilization facilities which was relatively expensive. The final economic indexes of the described foundry application were as follows:
- high decreasing of production costs about 20%,
- increasing of the cupola effectiveness about 50%,
- coke consumption decreasing about 10%.

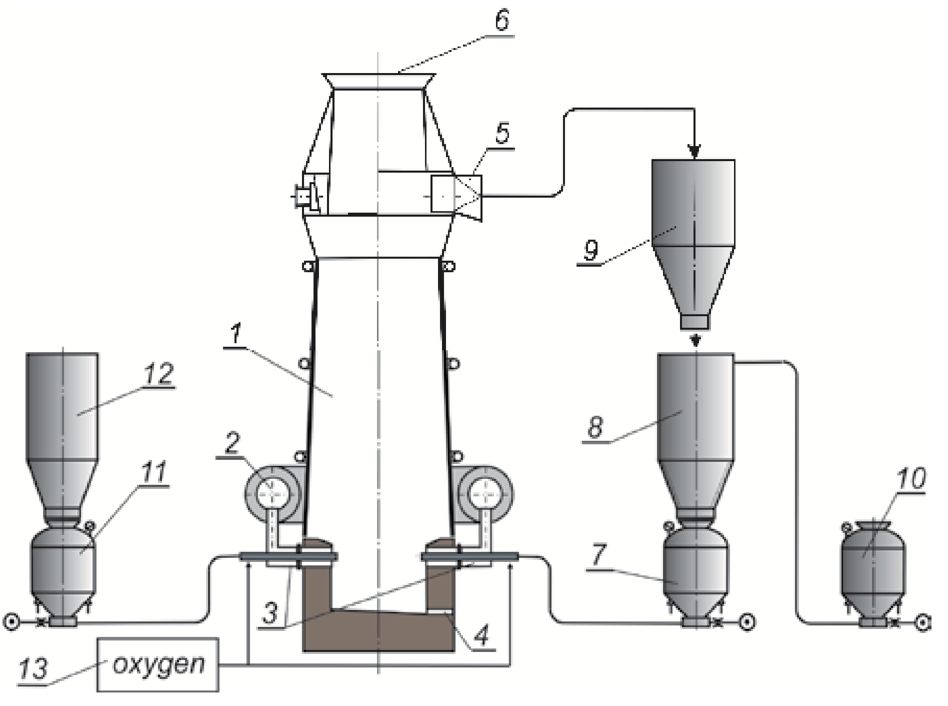

Fig. 3. Industrial cupola dust injection setup; 1- cupola, 2- wind box, 3- cupola nozzles (tuyeres), 4- tapping hole, 5- dust removal system, 6- charging door, 7- dust-coal mixture pneumatic chamber feeder, 8- dust-coal mixture container, 9- dust storage container, 10- coal pneumatic chamber feeder, 11- ferroalloys pneumatic chamber feeder, 12- ferroalloys storage container, 13- oxygen blow

2.3 Metallurgical furnace dust injection for slag foaming

One of the biggest problems in metallurgical and foundry industries is a large quantity of dust generated during production processes. The most important is furnace dust created when the molten metal is prepared and subsequently sucked out by the dust removal system (Machado et al., 2006, Ruiz et al., 2007; Vargas et al., 2006). At the Department of Foundry, the experiments with use of pneumatic injection method were carried out to utilize these kinds of materials and some results were successfully introduced into industrial applications. This part of the chapter shows that pneumatic injection technique could and should be continuously considered as an effective method for dust wastes utilization. The mass of dust generated during steel-making is enormous according to (Jezierski et al., 2008; Holtzer, 2005; Fiore et al. 2008). In Europe it is roughly 900 000 t/year, in Japan over 450 000 t/year and in Poland about 60 000 t/year. Over 30% of total steel production is molten nowadays in electric arc furnaces (EAF) and one of the most significant environmental issues is utilization of dusts, often with high zinc content. Back in 1990s the experiments were started worldwide with dusts re-injection into melting furnace. The Department of Foundry of Silesian University of Technology a few years ago carried out the researches and then industrial implementation of the installation for dusts pneumatic injection back into 65 tons EAF in one of the Polish steel plants. The goal was to utilize the furnace dust in mixture with pulverized coal what should be good for slag foaming. The EAF's slag foaming method is well known and successfully used as a necessary approach for economical electrodes use, energy management and stability from the melting process point of view (electric arc stabilization). The scheme of the slag foaming process with use of pneumatic injection technique for reagent's mixtures introducing were presented in Fig. 4 below. The mixture of furnace dust and pulverized coal in the ratio of 3 to 1 was prepared. It was both due to chemical and technical reasons, firstly, to ensure estimated carbon content to start physical and chemical foaming reactions and secondarily to ensure fast and stable pneumatic conveying of the material through pipeline and finally injection lance. The furnace dust alone causes problems during pneumatic conveying and may suspend inside the feeder.

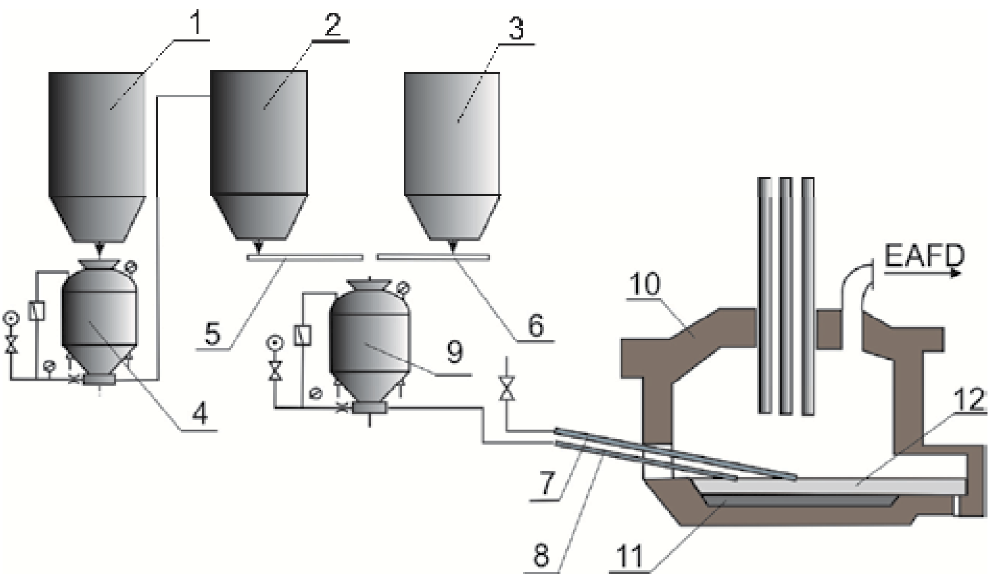

Fig. 4. The industrial set up for EAF dust-coal mixture pneumatic injection into 65 tons EAF: 1- furnace dust container (feeder), 2- intermediate dust container, 3- pulverized coal container, 4- furnace dust pneumatic chamber feeder, 5- dust feeding screw, 6- coal feeding screw, 7- oxygen lance, 8- mixture injection lance, 9- mixture pneumatic feeder, 10- EAF, 11- liquid metal, 12- expanded slag

The carrier gas (compressed air) was dried and the diphase gas-particles stream was created with use of the pneumatic feeder of own design (the former cooperation with POLKO company) and injection process started. The chemical composition of furnace dusts was typical for the steel plant which utilizes industrial scrap including automotive sheets with high zinc content. The researches were carried out firstly in laboratory conditions on the semi-industrial setup to estimate pneumatic conveying parameters for the material. After this stage the decision of producing a special chamber feeder design (with possible material fluidization) was made to ensure no material is suspended inside. Then the POLKO company (one of the best Polish companies in the pneumatic conveying field, formerly a part of the Department of Foundry) designed and manufactured the complete powder injection set-up: feeders, pipelines, automation and control devices etc. and the industrial experiments were started. During the industrial experiments a total of 278 melts were performed with various material compositions as follows:

- 167 melts with dust-coal mixture injection (90% of dust and 10% of powdered coal),
- 69 melts with dust-coal mixture injection (25% of coal and 75% of dust),
- 42 melts with the coal injection only (for comparison).

The main parameters of the process were:

- dust size: 0.005÷0.5mm,
- dust bulk density: 489kg/m^3,
- coal grain size: 0÷3mm,
- coal bulk density: 667kg/m^3,
- maximum mass of the mixture injected during one melt: 1330kg,
- mass composition of the mixture: 75% of dust + 25% of coal,
- mixture injection time: 10÷15min,
- system capacity: 0.5÷2.2kg/s,
- unitary oxygen consumption: 2÷4m^3/t,
- unitary dust consumption: 5÷11kg/t,
- unitary coal consumption: 1÷3kg/t.

The experiments proved high efficiency of the installation and after some minor parameters adjustments it was successfully commissioned and has been used till now. The energy consumption rate decreased significantly, the electrodes life extended and the process stability was improved, too. However, the most important result is that the plant utilizes all furnace dust generated by itself with several times less dust capacity deposited on dumps.

3. Sand reclamation

After the casting is knocked out the mould the used moulding and core sand become the by-products. The used sand can be utilized separately or with other components in the building industry, as a leak stopper on waste storage areas, as a material in concrete aggregate production and as a filler in a roads building industry. However, the important issue is to maximise utilization of used moulding sand by foundry plant itself by employing sand grains reclamation methods. The specific sand component recovery is a complex issue because the recovery alone is not enough to give the component the appropriate properties to use it again in new moulding sand preparation. The recovered components must meet quality requirements and be a fresh components replacement of full value (Danko, J., & Danko, R., 2004; Danko J. et al., 2007; Danko R., 2004).

The sand reclamation process consists of following actions:

- preliminary mechanical impurities (metallic ones mainly) from the used sand separation,
- agglomerated sand break-up after casting knock out,
- screening and proper grain size fraction separation,
- repeated metallic inclusions separation process,

- real sand reclamation, dry or wet, the goal is the residues of binding material removal out of sand matrix, with use of the methods which allow to remove thin material coating from the grain surface,
- sand matrix de-dusting or rinsing to remove all the unwanted reclamation products, proper grain size fraction of specific size and homogeneity separation (the classification on the grain size basis) (Szlumczyk 2005; Szlumczyk et al., 2007, 2008).

The field of sand reclaim application depends on the sand matrix grains cleanness degree that is binder from the grain surface removal and reclamation products classification. The essential reclamation process part is binder removal, that can be realized by abrasive sand matrix grains mutual reaction. The selection of the devices setup fitted for the reclamation process depends on the binder grade and the quality requirements for the reclamation products. The sand reclamation methods can be divided into wet and dry. In the second group the mechanical and pneumatic reclamation occur in the ambient temperature and thermal reclamation in the elevated temperature. In the wet reclamation method the used sand is mixed with water and in the form of pulp is mechanically treated usually in the rotary device. The sand grains are released not only from thin binder coatings and insoluble in water impurities but partly from insoluble impurities which can dispergate, too. The sand matrix after binder separation is rinsed, classified, dried and cooled.

In the mechanical method usually the machines are used which grind (mill), abrade or strike sand grains. In the pneumatic method which is a specific mechanical method modification, the binder layer removal is obtained by the collisions and abrasion of the sand grains in the air flow (cocurrently). In the pneumatic method the used sand conveying stream energy between technological appliances is employed. It is possible to insert the linear regenerator into straight segments of the installation which is purposely geometrically shaped (some throats are introduced) or on the pipe outlet to mount abrasive-percussive cap, which changes the stream direction. The controlled disturbance in pneumatic stream inside pipeline intensifies abrasive cleaning of the binder residues from matrix grains process. The movement of the pneumatically driven particles is defined by the resisting forces caused by gas and material friction on the pipeline's walls, particles friction on themselves and gravity and inertial forces of lifted particles. The reclamation process was carried out on the installation fitted to sand matrix pneumatic reclamation with the linear regenerator and abrasive-percussive cap (Szlumczyk 2005; Szlumczyk et al., 2007, 2008). The experimental reclamation setup consists of the following systems (Fig. 5):

- high-pressure pneumatic conveying chamber feeder (1),
- linear regenerator (3) cooperating with the pipeline of D_N=0.08m diameter,
- receiver (4) connected to de-dusting system,
- fluidized air classifier (6) which is a separate device,
- gas mass flow meter (5),
- abrasive-percussive cap as a receiver (7),
- controlling, measuring and regulating instruments.

The linear regenerator parts used in the experiments were shown in Fig. 6. They were made of wear resistant plastic and their shape corresponds to Witoszynski nozzle on inlet and Laval nozzle on outlet and such a shape was a result of model experiments described further in the paper. The analysis of the receiver (abrasive-percussive cap) influence on the binder coating removal (destroying) process correctness can be based on the maximum air flow rate. The cap was shown in Fig. 7. According to the stream theory and own research it can be assumed that for the distance between pipeline outlet point and abrasive-percussive cap up to l_1=0.3m the particles of d_s>0.3mm diameter, velocity does not change. To simplify: for the manufacturing method sake it was set up that the inner cap surface shape is a cylinder of R_1 radius. On the basis of the experiments results as well as calculations it can be stated that the pneumatic sand matrix reclamation installation is suitable for the sand grades being examined. The effectiveness of the linear regenerator depends on compressed air supply system parameters what is essential to achieve proper diphase stream parameters. These parameters are transportation velocity and mixture mass concentration. The significant element of the proper process run is the constructional design of the throat. It is decisive for the resistance of flow. When the throat degree is small, the process efficiency decreases while for a too large one (over S_p=4) the resistance increases what makes it impossible to achieve better efficiency and more than one use of the throat elements on the sand matrix being reclaimed stream way. The carried out experiments indicated that the best results of the linear regenerator application were obtained for the flow of w_8 velocity from 15 to 28m/s and μ_m=12 to 25kg/kg mixture mas concentration. In these conditions the system ensures good sand matrix reclamation process results for the moulding sand being processed. The use of the abrasive-percussive cap needs the diphase stream velocity on the pipeline inlet into cap adjusting.

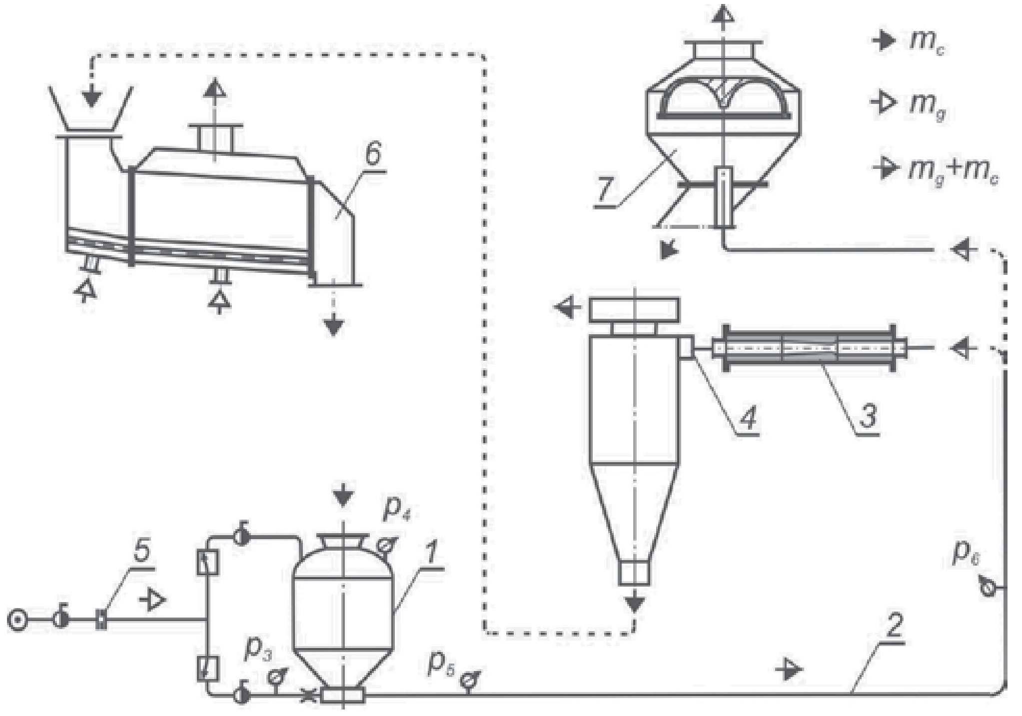

Fig. 5. The experimental setup scheme

The velocity should not exceed critical value what may cause sand matrix grains deterioration (cracking and scaling). The acceptable velocity of the stream introduced into cap w_{AN} = 35 m/s. When the effectiveness of these two pneumatic sand matrix reclamation systems is analysed, it can be stated that they are more beneficial than other dry reclamation systems.

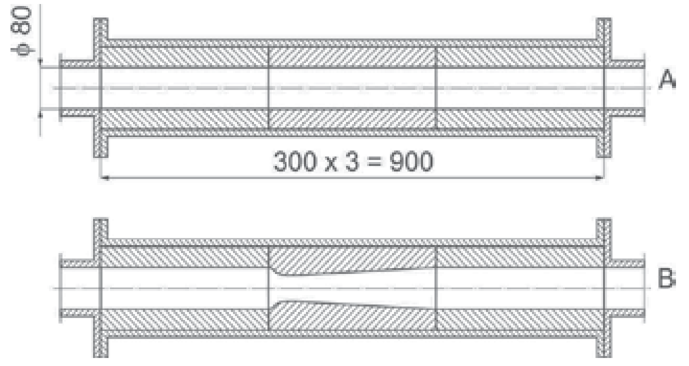

Fig. 6. The linear regenerator parts

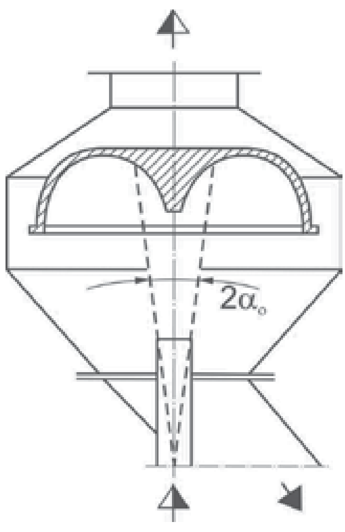

Fig. 7. The scheme of the stream on the abrasive-percussive cap surface influence process.

4. Physical modelling of the powder injection process

The observation of the diphase stream is often impossible. The conditions that limit direct observation are high liquid metal temperature (in powder injection into liquid metal process) or dustiness (in the sand reclamation process). Therefore physical modelling experiments are carried which allow to some extent to explain the phenomena visible in diphase stream conveying processes. The experiments on the models must be carried out with regard to the similarity theory otherwise the results cannot be transferred onto industrial installations and can be analysed in only exact research conditions. There are several similarity conditions description methods i.e. relationships between physical quantities scales which describe some phenomenon being examined. All of them are based on the dimensional analysis (Clift et al., 1978; Farias & Irons, 1986; Sawda & Itamura, 1989).
Many authors dealt with the issue of the diphase stream (Szekely, 1979; Zhang, & Fruehan, 1991; Zhao &Irons, 1990). Most often the model experiments are conducted on the various liquids, gases and solids being introduced. The results estimated with the help of the criteria

number are transferred on the liquid metal conditions. Increasingly, for analysis of these parameters, numerical modelling and computer simulation of the occurring phenomena is conducted after previous physical modelling of the powder injection into liquid process been made. During the observation of gas or gas and solid mixture flow introduced into the metal bath almost every scientist distinguished two flow states: bubbling (so-called barbotage) and jet flow. The first is characteristic for the small material mass flow and velocity on the lance outlet. The mass transport occurs only on bubbles surface, which are deformed and disintegrated just under the surface of the liquid medium where they are introduced. The second condition is characteristic for the big material mass flow and velocity on the lance outlet. The large bubbles deform and disintegrate just on the lance outlet that causes the large reaction surface between liquid and solid material being introduced. This condition is much more beneficial than the barbotage. For small injection velocity the bubbles break away the stream momentarily. When the velocity is higher the stream penetrates the liquid further and wrinkles and the small bubbles appear. The stream introduced into liquid causes the injected material with liquid mixing and the assimilated droplets transport the stream further. When the stream velocity increases the larger gas amount mixes with the liquid (Janerka et al., 2004).

The goal of the physical modelling is sometimes the introduced diphase stream surface estimation what is an area of the intense mass transport between solid reagent and metal bath and the stream penetration range. The aim of the experiments is to show what parameters and how significantly they influence the shape and size of the diphase stream area inside liquid medium. Such experiments are carried out on the special setups for the physical modelling. The example of the setup based on the high pressure pneumatic conveying chamber feeder is shown in Fig. 8. The material supplier is the pressure container (1) of 3.0dm^3 capacity.

The closing valve is mounted at the top of the container. The overpressure inside the container which device efficiency is based on, is regulated by means of the reducing valve (7). The spring-type pressure gauges in particular setup places were mounted to measure the overpressure. The carrier gas supplying system consists of compressor (8), cut-off valve (9) and the reducer with filter (10). The gas flow meter (11) was used to gas flow measuring. The powders introducing systems contain pipes (12) ended with a lance (13) introduced into

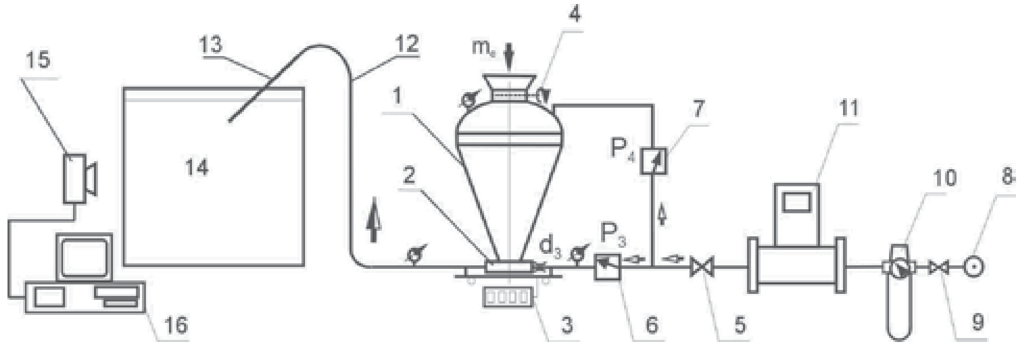

Fig. 8. The example setup for the physical modelling: 1-pressure container, 2-mixing chamber, 3-scales, 4-closing valve, 5,9-cut-off valves, 6,7-pressure reducers, 8-compressor or pressurized argon bottle, 10-carrier gas filter, 11-flow meter, 12-pipes, 13-injection lances, 14-model liquid container, 15-digital camcorder or camera, 16-computer

container (14) and made of Plexiglas (dimensions 1000x500x100mm). Every experiment is recorded on the digital camera (camcorder) and the captured pictures are transferred to the computer. As a model, powders of water insoluble materials are mostly used (plastics). Such experimental example with the use of polystyrene and polyethylene of 0.26-1.20mm diameter was presented below. The injected materials density varied from ρ = 822÷1240 kg/m^3. As a model, liquid water and water NaCl solution of 1180 kg/m^3 density were used. The injection process was carried out with the injection lance of 5mm diameter and it was sloped at an angle of α=30, 45, 60° to the liquid surface and submerged to depth h=50 and 100mm. The air only injection into liquid medium with various velocities was shown in Fig. 9 while in Fig. 10 next page the diphase stream injection with various parameters was presented.

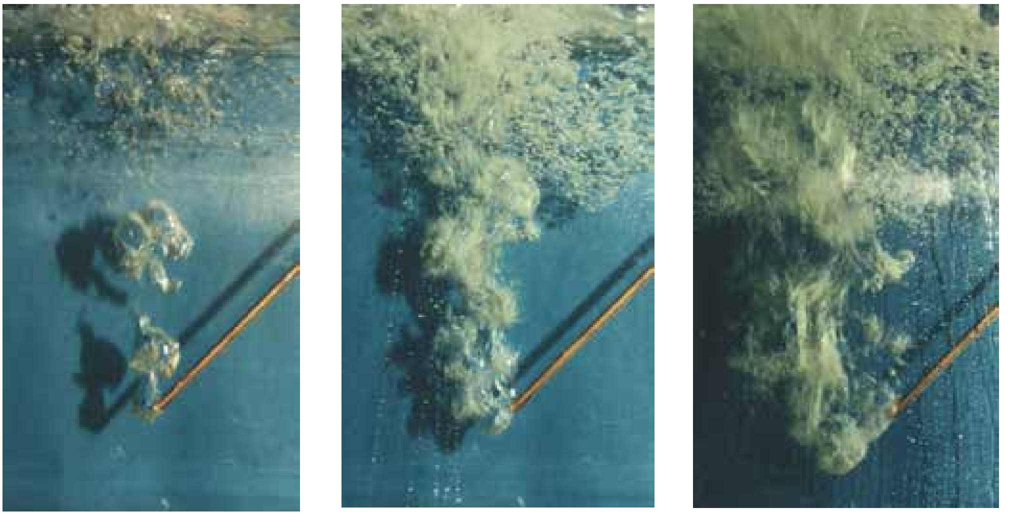

Fig. 9. The single phase (the air) injection into liquid medium with velocity of w =6.8 m/s, w =37.1 m/s and w =78.5 m/s

The single phase stream penetration range increases as the carrier gas mass flow increases. However, it is several times smaller than for the diphase stream injected under the same pneumatic conveying parameters. It is because the higher diphase stream energy is mainly kinetic. The diphase stream injection causes fewer disadvantageous phenomena appearance on the liquid surface (splatters). The higher stream penetration range is also obtained when small particles are introduced. It may occur because the smaller particles present higher velocity on the lance outlet and the liquid medium resistance for these particles is less. This is a good condition (from the process point of view) because smaller particles give larger extended surface of the injected powder for the same total volume of the injected particles. It results in higher technological indexes such efficiency and recarburization rate. However, fine powders of small density cause problems during pneumatic conveying because of their tendency to go into suspension inside chamber feeders and non-uniform falling down the container.

The diphase stream area can be divided into four characteristic zones (Fig. 11).

Zone I – close to the lance outlet. In this area large gas bubbles of irregular shape are created. Their size and number depend on gas flow. When the flow is higher, bubbles break away from the lance faster and faster disintegrate and again new bubbles are created. The carburizer particles are captured in them and after the bubbles burst they will have a contact with liquid metal. However, it occurs close to or even on the metal surface. The mass exchange takes place as a result of metal movement and carburizer grains floating on the metal surface. This is a disadvantageous phenomenon but it can be minimized with the gas velocity on the lance outlet increase.

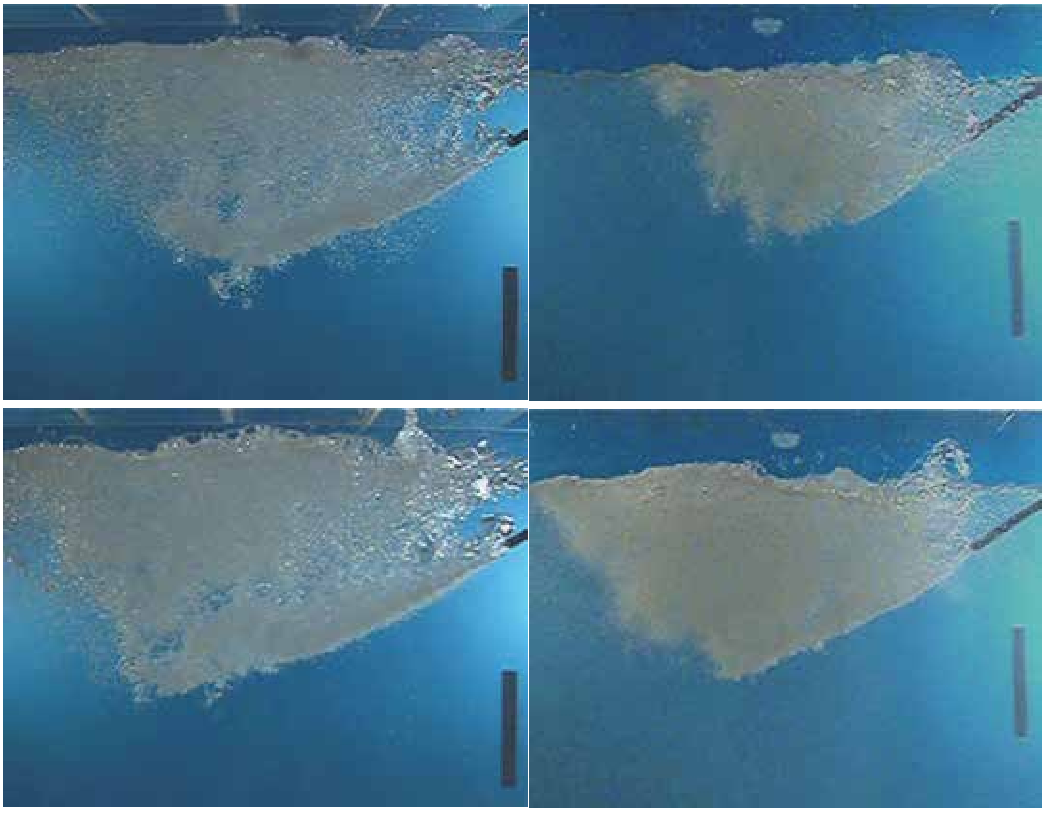

Fig. 10. The diphase stream injection for the particles of d_c=0.833 mm diameter and ρ=882 kg/m^3 density, liquid density ρ_{osr}=1000kg/m^3 (left) and ρ_{osr}=1180 kg/m^3 (right picture)

Zone II is a direct stream range area. It mainly consists of carburizer particles because only they have enough energy to infiltrate the liquid metal so deeply. The mass exchange process is the most intense in this zone because the particles have significant velocity so the near-surface diffusive layer thickness is very small.

Zone III is the area of the smallest particles having direct contact with liquid metal. Its area is the largest and it may be assumed that it determines the process efficiency. The size of this zone is a consequence of zone II creation.

Zone IV consists of bubbles of spherical, ellipsoidal or spherical cap shape, dependably on the bubble creation place and its size. Moving towards the surface the hydrostatic pressure decreases what causes their growth. Being in liquid metal they heat themselves additionally and their volume increases. They burst close to or on metal surface so the particles are there partly uplifted. The carburizer particles are captured inside these bubbles and the mass exchange occurs after their bursting under the liquid metal surface what significantly decreases the process total efficiency.

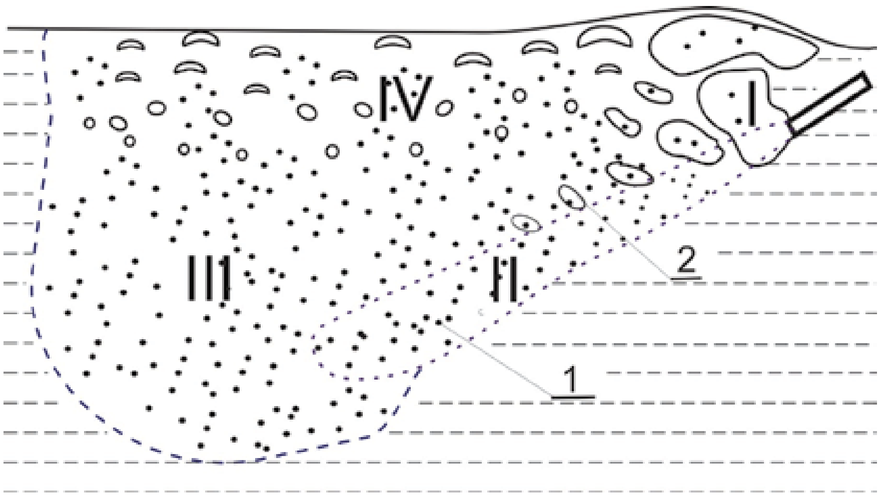

Fig. 11. The shape and area of the diphase stream: 1-carburizer particles, 2-gas bubbles

Under industrial conditions of pneumatic recarburization the estimated process efficiency is obtained thanks to overpressure and exchangeable nozzles in dosing device changes. It allows controlling gas flow as a one of the main parameters of pneumatic conveying process. The dosing device output increase is obtained mostly by increasing overpressure inside chamber feeder container. Subsequently the flow increase (mass gas and material flow) causes adequate surface area, width and penetration range of the diphase stream increase. The fine particles injection is very beneficial not only from metallurgical point of view (large contact surface between reacting phases) but because the more significant diphase stream surface and direct reaction zone metal-carburizer increase, too.

The model experiments were carried out to select the best geometrical layout of the throats used in the linear regenerator, too. The research consist of stream flow conditions analysis in various geometrical throat layouts inside pipe system. The aim was to force the flow instability that causes mutual particle interaction. During the experiments the stream flow of various mass concentration velocities and the pressure drop on the measured section of the pipe system were recorded. The particles distribution inside this stream was analysed.

These experiments were also recorded photographically. Typical photographs of the solid particles distribution inside diphase stream were presented in Fig. 12.

The model experiments were employed to optimize constructional setup of the throats in the linear regenerator and to estimate their shape (Witoszynski nozzle on the inlet and Laval nozzle in the outlet). These sections were made of transparent material (Plexiglas) to make particles movement observation during the diphase stream flow possible. The solid particles were granulated polypropylene ones of 2-3mm size and black and white colours. The recordings and observations results allow analysing the diphase stream flow parameters inside the particular linear regenerator sections. The particles agglomerations are visible on the linear regenerator inlet (throat) what suggests their mutual interactions intensity increase with the small resistance of flow caused by differential pressure.

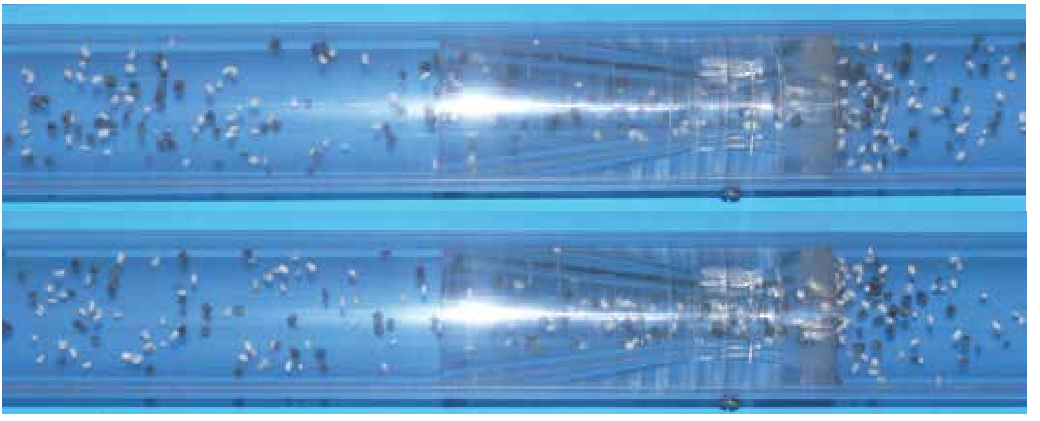

Fig. 12. The stream flow inside model linear regenerator system

4.1 Diphase gas-particles stream force model analysis

The experiments (as a continuation of the model experiments described above) were carried out to understand the character of diphase stream forces on liquid surface in powder injection process. The short description of work methodology and apparatus are mentioned in the paper as well as the examples of the results obtained. The work presented in the paper is a part of a large scaled experimental plan that should explain important relations between injection technological indexes and dynamics of the diphase stream. The research stand is presented on fig. and its complete description was presented in previously published paper (Jezierski et al. 2006) but instead of furnace or ladle a measuring device is situated at the end of the injection system and connected to PC computer, see Fig. 13.

The experiments were conducted as part of the experimental plan for various lances geometries, pneumatic parameters and injected powdered materials sorts. Use of PC computer with dedicated program allowed to measure stream force value with frequency 10 measurements per second. So we can say that the stream force measurement was almost continuous. The powdered material used in the experiment was polystyrene with granulation 0.4mm with the air as a carrier gas. The distance between lance outlet and an extensometric measuring device's surface was established at three levels: 10, 40, 80mm because one of the problems to solve was that distance influences stream force's value achieved.

The full experimental plan included 27 experiments for various process parameters configurations separately made. Apart from a grain size there were four another independent variables during experiments:

- a carrier gas (compressed air) pressure p_1, (three levels of changing: 0.1; 0.2; 0.3 MPa),
- a gas into dispenser pressure p_4 (six levels of changing: from 0.05 to 0.3MPa with step 0.05MPa),
- a distance between lance outlet and measuring device surface H (10, 40 and 80mm),
- a lance inside diameter d_w, (three levels of changing: 5.6; 6.1 and 7.6mm).

The results of the recordings and calculations were used to analyze and to create the graphs to show time-changing character of stream force. The examples of the graphs for experiments with use of lance with inside diameter 6.1mm were presented below in Fig. 14. One can see a characteristic peak at the end of the blowing. It is connected to moment when the last portion of mixture is blown through the injection lance. From technological point of view the most important is the period when force stabilizes in the middle of the cycle because in real industrial conditions we are interested mainly in the process stability. When one looks precisely at graphs one can see that for some combination of pneumatic parameters p_1 and p_4 quite considerable stream force fluctuations can be seen. It is mostly present in cases when the pressure into container (above powdered material) p_4 has value from the highest levels equal 0.25 or 0.3MPa and carrier gas pressure p_1 has the smallest value equal 0.1MPa (Fig. 15 next page). In such conditions for small lance's inside diameter the mass concentration of diphase mixture value is too big so the pneumatic conveying character seems to be pulsating not stable.

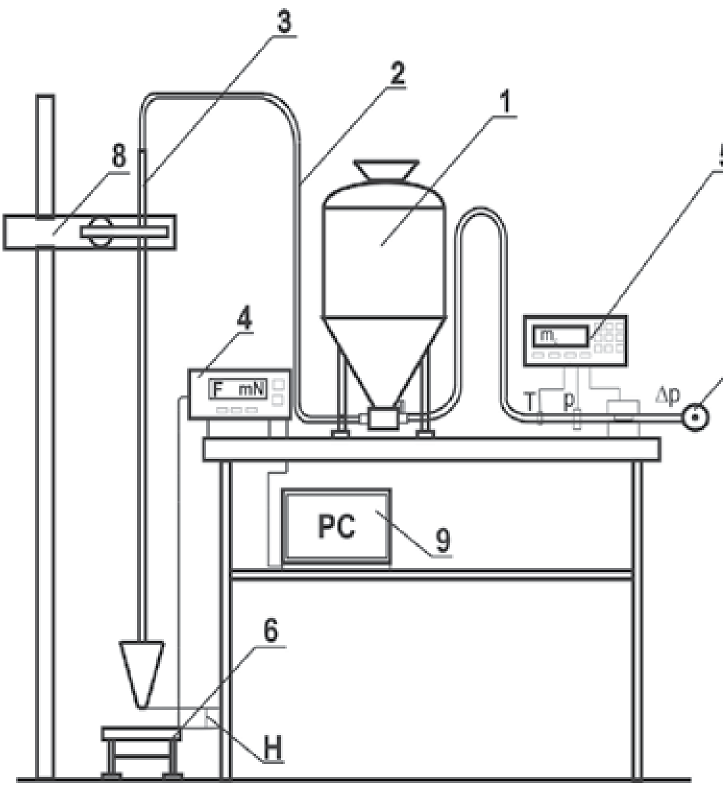

Fig. 13. Scheme of research setup; 1- pneumatic powder chamber feeder, 2- pipeline, 3- injection lance of special design, 4- stream force measuring electronic device, 5- carrier gas flow meter, 6- extensometric device, 7- carrier gas (compressed air) supply, 8- slidable arm, 9- PC computer, H- changeable distance between lance outlet and measuring device

The paper presents graphs for only one chosen inside diameter lance d_w = 6.1mm but the described problems and relationships between process parameters were present in others examined lances, too. The fluctuations of force values were the biggest with use of the smallest lance of 5.6mm diameter and the most stable process was observed for the lance of 7.6mm inside diameter.

The next step was statistical analysis of recorded and calculated data. The average value of stream force in stable (during the stable cycle period) was calculated, the experimental equations were formulated and graphs were made. Below are presented some of them for the parameters analogical to these on the stream force's time-changing graphs, see Fig. 16 and 17 next pages.

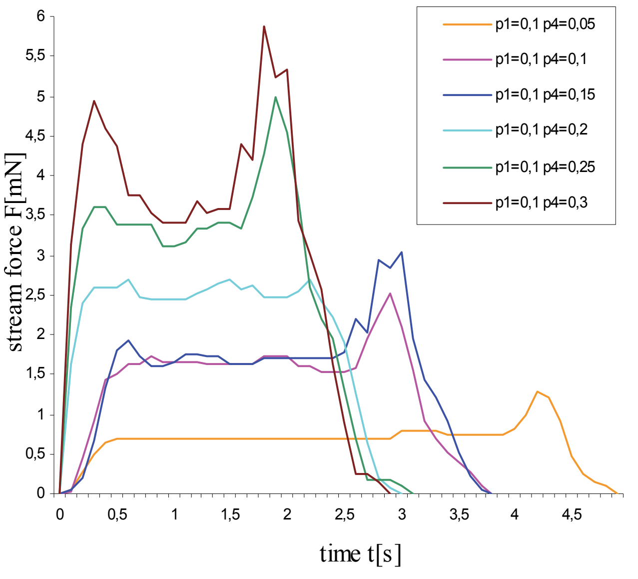

Fig. 14. Diphase stream force character for parameters as follows: lance diameter d_w = 6.1mm, distance between lance outlet and measuring device's surface H = 40mm, carrier gas pressure p_1 = 0.1MPa, powdered material – polyethylene of granulation 0.4mm

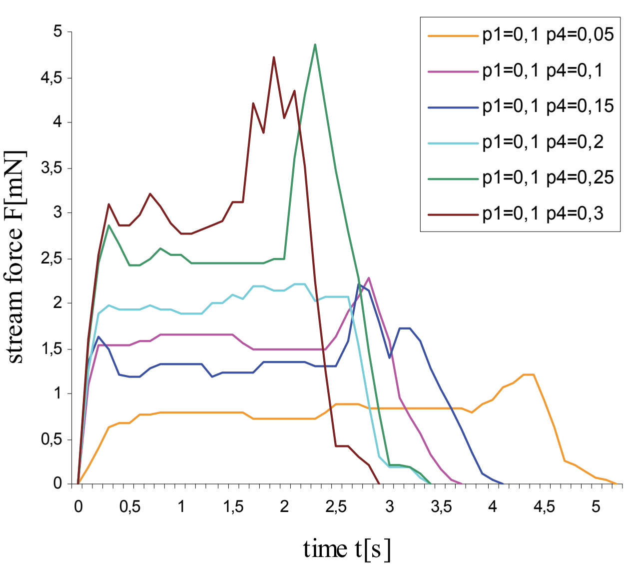

Fig. 15. Diphase stream force character for parameters as follows: lance diameter d_w = 6.1mm, distance between lance outlet and measuring device's surface H = 80mm, carrier gas pressure p_1 = 0.1MPa, powdered material – polyethylene of granulation 0.4mm

$$F = -0{,}785 + 0{,}032 \cdot w_k + 0{,}019 \cdot \mu \tag{2}$$

where: w_k – gas velocity, μ - mass mixture concentration.
The described experiments have drawn to the following conclusions:

1. Velocity of the carrier gas in the lance outlet depends mostly (the same geometrical conditions) on inside lance diameter and mostly influence diphase stream force value.
2. Diphase stream force value increases with increasing pressures (especially pressure in powder feeder p_4 which increase cause mass concentration μ increasing) and decreases with increasing of distance between lance outlet and measuring surface (liquid metal bath). For distances above 40mm the value was so small that it was impossible to measure with used equipment.
3. The proper period of injection cycle for industrial conditions is in the middle of the process, when the stream force has good stability. The moment which show the finish peak introduced quite considerable amount of carrier gas with last portion of powder injected.
4. Mass concentration of the diphase mixture and velocity of carrier gas in the lance outlet have decisive influence on the analyzed force. But the value of μ should not be above 20-30kg/kg because the higher values cause high instability of conveying process.

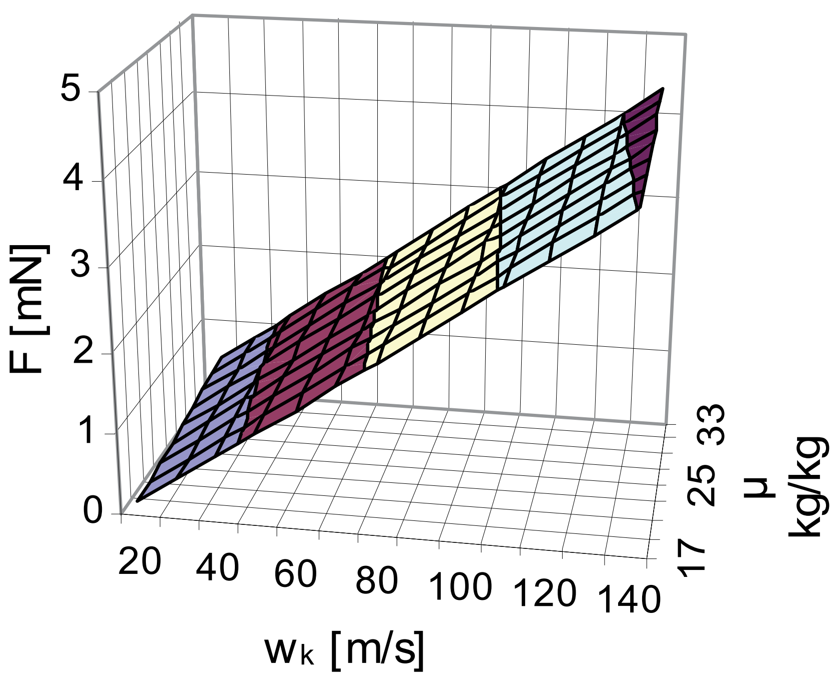

Fig. 16. Influence of gas velocity in lance outlet and mass concentration on the stream force (d_w = 6.1mm, H = 10mm)

The further model experiments were carried out with the liquid medium and they proved the previously made researches without liquid usage. The stream force value corresponds strongly with the ability of the stream to infiltrate the liquid with the high stream penetration range.

Below (fig. 17) there is shown a dependence between diphase stream force and pressure values.

$$F = -0{,}063 + 0{,}809 \cdot p_1^2 + 1{,}831 \cdot p_4^2 \tag{2}$$

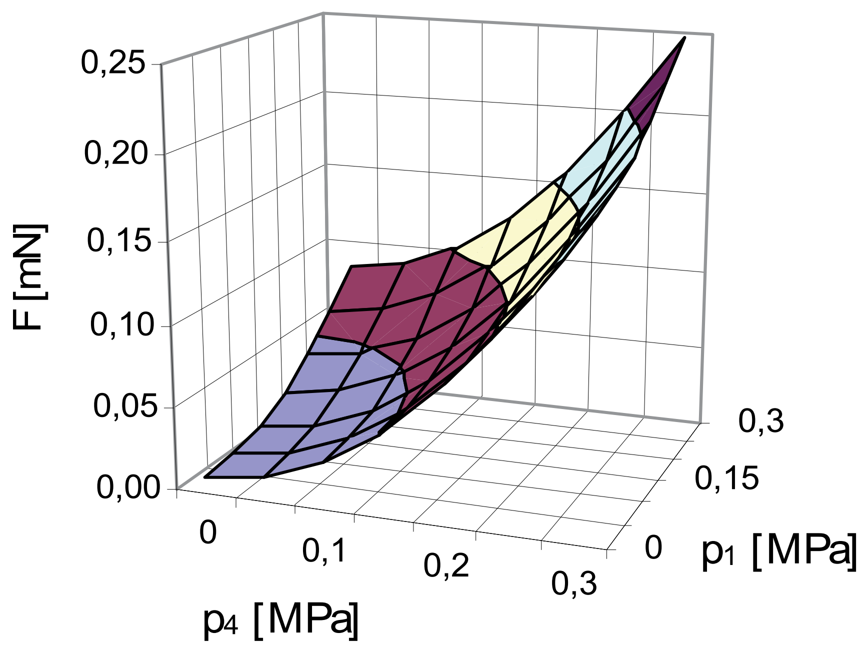

Fig. 17. Pressure p_1 and p_4 values influence the stream force (d_w = 6.1mm, H = 10mm).

5. Conclusions

In the chapter the usage of pneumatic powder injection method for solid metallurgical and foundry wastes mainly in form of powder or dust utilization was briefly presented. The experiments in this field have been made in the Department of Foundry, Silesian University of Technology for many years. These several experimental examples as well as industrial applications show how this technique can be employed to utilize furnace dusts generated in various kinds of furnaces, fine ferroalloys fractions (by-product of lumpy ferroalloys production) into liquid metal bath introduction, liquid ferrous alloys (mostly cast iron) recarburization and used moulding sand reclamation (pneumatic method). The results of author's experiments proved the high effectiveness of this method in every of the mentioned processes. The ecological and economic parameters of industrial application are very promising so the interest of the industry continuously increases. It seems to appear especially significant nowadays when environmental protection is one of the most important problems and when pollution limits are very low, too.

To develop further the theory of diphase stream movement and its characteristic inside liquid medium from the powder injection process point of view the next researches have been just launched. Their goal is to examine stream flow with use of the high speed camera to catch the real particular powder particle movement. Both model experiments and real injections into liquid metal are planned to observe how the gas-powder stream really enters the liquid metal surface. It will be the first such approach to the pneumatic powder injection process and the results of the experiments will be published later.

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Economic and Operational Feasibility Analysis of Solid Waste Minimization Projects

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1. Introduction

The purposes of this chapter are to demonstrate a structured process to evaluate and determine the operational and economic feasibly of solid waste minimization projects that are based on proven financial engineering concepts. Many organizations are concerned with reducing solid waste levels, but few have the tools and necessary resources to evaluate and select among competing projects. These projects can range from fixed equipment purchases, such as balers or digesters, to implementing an office recycling program. This chapter provides a standardized business-based process to evaluate and select among competing solid waste minimization projects to determine which will best meet the organization's goals and maintain compatibility with existing processes. The analysis process involves identifying the benefits, costs, and drawbacks associated with each alternative project. To accomplish this, each alternative is evaluated based on: the impact on the program goal, technical feasibility, operational feasibility, economic feasibility, sustainability, and organizational culture feasibility. As a companion, a case study from Lucas County, Ohio (USA) is provided that demonstrates the analysis process. In addition, the paper explores the impact of uncertainty in decision making by highlighting economic efficiencies, sensitivity analysis, and changes to the data inputs, specifically inflation, recycling levels, and recycling commodity market shifts. This chapter may serve as an example or model for organizations considering the implementation of competing solid waste minimization projects.

2. Screening alternatives

The process of identifying waste minimization alternatives can generate numerous options. It would be very time consuming for the team to conduct a detailed financial and operational feasibility evaluation for each option. A quick screening process can help to rapidly identify the options worthy of full evaluation and the possible inclusion in the waste minimization program. Additionally, non-effective options can be removed, saving the team valuable time and money in the evaluation process. An effective screening process should be based on the original goals of the project and at a minimum should examine:

- The expected solid waste reduction (tons per year)
- The expected start up costs
- The impact to waste removal costs ($ per year)

- The impact to purchasing costs ($ per year)
- The impact on employee moral
- The ease of implementation

The team should keep in mind that the goal of the screening the process is to quickly identify options worthy of further analysis. A weighted scoring system should be developed and applied to rank each alternative in an objective manner. A Quality Deployment Function, such as the 'House of Quality' is an excellent tool to accomplish this evaluation. A House of Quality is a graphic tool for defining the relationship between the organization needs and the capabilities. It utilizes a planning matrix to relate the organizational wants (for example solid waste reduction and cost performance) to how the waste minimization program will or can meet those wants (for example process changes or recycling efforts). It looks like a house with a correlation matrix as its roof and the organizational wants versus waste minimization options as the house structure. Another benefit of the House of Quality is that is may increase cross functional integration within the organizations using it, especially between marketing, engineering, and manufacturing.

Before proceeding with the screening process, the team should decide on the evaluation criteria (the "What's") and weighting system. A scale of 1 – 10 for weighting each criterion is recommended. These weightings should be determined by the team, project manager, facility manager or a combination. The evaluation criteria should be directly related to the overall goals of the project, such as:

- Reduction in waste amounts
- Reduction in waste toxicity
- Reduction to waste disposal costs
- Reduction in purchasing costs
- Revenue generation potential
- Low start up costs
- Productivity improvements
- Quality improvement
- Ease of implementation
- Impact on employee morale
- Impact on organization image
- Impact on safety
- Other factors as determined by the team

Once these criteria have been created, the team should rank them on a scale of 1 to 10 based on importance. For example, regulatory compliance of each option may receive an importance rating of 10 (meaning that it is highly important). On the other hand, a criterion such as low start up costs may receive an importance rating of 2 (meaning that start-up costs are of low importance and not a major concern in the decision-making process).

Once the criteria and importance ratings have been established, the team should list each alternative in the rows of a spreadsheet. In the row for each alternative, the team should place a rating score corresponding to the level of which the alternative meets the criterion with 0 being no or very low impact and 10 representing great impact. For example, if the team is considering the purchase of a cardboard baler, the reduction in waste amounts could be significant, so the team may rate it a an 8, but in the start up cost criterion, the team may rate it lower, such as a 1, due to the high implementation cost to purchase the baler. Once each alternative is rated, the ratings should be multiplied by the importance factor and each row should be summed. This score will allow the team to objectively screen each alternative. Once all of the alternatives are listed and scored, the team can screen them based on there total score. Alternatives with higher total scores pass the screening process and become eligible for further evaluation. To determine the cut-off point, several methods may be applied that depend on time and money resources. For example, the team may set the minimum threshold at a specific point value, accept the top 20%, or accept the top ten for further analysis. When first starting a solid waste minimization program it is recommended that the team select the top third (33%) of all alternatives for further screening to compensate for estimation errors.

3. Analyzing and selecting alternatives

After reducing the list of alternatives using the screening process discussed in Section 1, the remaining alternatives should be further analyzed to determine the best fit for the organization to minimize solid waste and hence include in the program. The analysis process will identify the benefits, costs, and drawbacks of each alternative. To accomplish this, each alternative is evaluated based on:

- The impact on the program goal
- Technical feasibility
- Operational feasibility
- Economic feasibility
- Sustainability
- Organizational culture feasibility

The key outcome of this phase is to fully document, analyze, and arrive at a final acceptance decision for each alternative. To accomplish this outcome, the process flow charts are analyzed; the annualized amount of solid waste generated is determined; a complete feasibility analysis is completed (including technical, operational, organization), a cost justification study is conducted; feedback is collected and analyzed; and finally a decision is made regarding each alternative (to implement or not implement). These studies provide a complete discussion and documentation for each alternative that will be used in the implementation phase if the alternative is accepted for implementation. During this process, the team must keep a clear understanding of the overriding goals of the waste minimization project. For example, the relative importance of reducing costs versus minimizing environmental impact. Some alternatives may require extensive analysis, including the need to gather additional data from vendors or to analyze market trends for recyclable material commodity markets. The first consideration when evaluating alternatives is its impact on the goals of the project established in the first phase of the project. These goals can range from in solid waste generation to the cost benefits associated with waste minimization. Efforts should first be made to reduce waste generation, next to reuse waste materials, next recycling (in and out of process) and finally disposal in a landfill. The idea behind the hierarchy is to engineer methods to eliminate the generation of a waste stream altogether and hence eliminate the need to manage the solid waste stream via recycling or landfill disposal. Alternatives should be separated into different categories to aid with this process. The categories are (based on the solid waste management hierarchy):

- Waste prevention alternatives
- Reuse alternatives
- Recycling alternatives

- Composting alternatives

The evaluation process itself, consists of seven steps to study each alternative. The process is completed sequentially and after each step, the alterative is accepted and 'moves' to the next phase or is rejected and the analysis is terminated without further steps being completed. If the alternative does not meet thresholds or feasibility tests, it is eliminated from further review to save the team time and resources. The alternative should still be kept on file in the event that technology or organizational changes render the option feasible. The seven steps are listed below:

1. Fully describe each alternative in terms of the equipment, raw material, process, or purchasing additions or modifications
2. Calculate the annualized waste reduction impact in terms of tons per year and whether the alternative is related to source reduction, reuse, or recycling
3. Compile and analyze the process flow charts that created the waste stream
4. Conduct a feasibility analyses (technical, operational, and organizational)
5. Conduct a cost justification for each alternative (payback, internal rate of return, and net present value)
6. Gather feedback from all stakeholders (internal and external)
7. Gain approval and sign off from the waste minimization team and organizational executives

3.1 Technical and operational feasibility

Technical and operational feasibility are concerned with whether the proper resources exist or are reasonably attainable to implement a specific alternative. This includes the square footage of the building, existing and available utilities, existing processing and material handling equipment, quality requirements, and skill level of employees. During this process, product specifications and facility constraints should be taken into account. A typical technical evaluation criterion includes:

- Available space in the facility
- Safety
- Compatibility with current work processes and material handling
- Impact on product quality
- Required technologies and utilities (power, compressed air, data links)
- Knowledge and skills required to operating and maintain the alternative
- Addition labour requirements
- Impact on product marketing
- Implementation time

When evaluating technical feasibility, the facility engineers or consultants should be contacted for input. In addition, it is also wise to discuss the technical aspects with the workers directly impacted by the change such as production and maintenance. If an alternative calls for a change in raw materials, the effect on the quality the final project must be evaluated. If an alternative does not meet the technical requirements of the organization, it should be removed from consideration. From a technical standpoint, the three areas that require additional evaluation are:

- Equipment modifications or purchases
- Process changes
- Material changes

If an alternative involves an equipment modification or purchase, an analysis for the equipment should be conducted. The team should investigate whether the equipment is available commercially and gather contact information/data from the manufacturer. Performance of the machine should also be addressed, including cost, utility requirements, capacity, throughput, cycle time, required preventative maintenance, space requirements, and possible locations in the facility that the equipment could be installed. In addition, if production would be affected during installation, this should be evaluated as well. The vendor or manufacturer may provide additional information regarding potential shut downs or delays. Required modifications to workflow or production procedures should be analyzed and any required training or safety concerns related to the equipment purchase should be reviewed. From an operational standpoint, attention should be given to how the alternative will improve or reduce productivity and labour force reductions or increases.

If a waste minimization alternative involves a process change or a material change, the impacted areas should be identified and feedback should be gathered from the area managers, employees, maintenance, and engineers (if applicable). With process changes, training requirements should also be discussed and determined. Also, the impacts on production, material handling/storage, and quality should be addressed. A material testing program is highly recommended for new items that the engineering team may not be familiar with so that they can analyze the impacts to quality and throughput. A design of experiment (DOE) that tests the changes versus the current material is an excellent method to gauge impacts. A DOE is the design of data gathering tests in which variation is present. Often the experimenter is interested in the effect of some process or intervention, such as using a new raw material, on some outcome such as quality.

3.2 Economic feasibility

From an economic standpoint, traditional financial evaluation is the most effective method to analyze alternatives. These measures include the payback period, (discounted payback period), internal rate of return, and net present value for each alternative. If the organization has a standard financial evaluation process, this should be completed for each alternative. The accounting or finance department or the organization should have this information. To perform these financial analyses, revenue and cost data must be gathered and should be based on the expectations for the alternatives. This may be complicated, especially if a project will have an impact on the number the required labour hours, utility costs, and productivity, or require initial investments or start-up funds. A comprehensive estimation of the cost impacts (revenues and costs) per year over the life of the alternative is required to begin the analysis. The first step of the economic evaluation process is to determine these costs. These costs include capital costs (or initial investment), operating costs/savings, operating revenue, and salvage values for each waste minimization alternative.

Capital costs are the costs incurred when purchasing assets that are used in production and service. Normally they are non-reoccurring and used to purchase large equipment such as a baler or plastic grinder. Capital costs include more than just the actual cost of the equipment; they also include the costs to prepare the site for production. Following is a brief list of typical capital costs; also know as the initial investment:

- Site development and preparation (including demolition and clearing if needed)
- Equipment purchases including spare parts, taxes, freight, and insurance
- Material costs (piping, electrical, telecommunications, structural)

- Building modification costs (utility lines, construction costs)
- Permitting costs, building inspection costs
- Contractor's fees
- Start up costs (vendor, contractor, in-house)
- Training costs

After the initial investment has been calculated, the reoccurring costs, savings, and revenues from the waste minimization alternative must be determined. The concept is to reduce waste disposal and raw material costs based on the implementation of the alternative that is being analyzed. For example, if a company considers the installation of a cardboard baler, the annual operating costs of the baler (such as labour and utilities), the annual cost savings from reduced disposal costs, and the revenue from the sale of the baled cardboard must be considered. Reducing or avoiding present and future operating costs associated with solid waste storage and removal are critical elements of the solid waste minimization process. Due to increased solid waste disposal costs (in the range of $30 - $80 per ton in the US); many companies are finding that the cost of waste management has become a significant factor in their cost structures. Some common reoccurring costs include:

- Reduced solid waste disposal costs – waste generation is reduced or is diverted to recycling streams resulting is less waste is being sent to the landfill for disposal and lower hauler charges. These include disposal fees, transportation costs, and predisposal treatment costs.
- Input material cost savings – options that reduce scrap, reduce waste or increase internal recycling tend to decrease the demand for input materials
- Changes in utility costs – utility costs may increase or decrease depending on the installation, modification, or removal of equipment
- Changes in operating and maintenance labour/benefits – an alternative may increase or decrease labour requirements and the associated benefits. The may be reflected in changes in overtime hours or in the number of employees.
- Changes in operating and maintenance supplies – an alternative may result in increased or decreased operating and maintenance supply usage.
- Changes in overhead costs – large projects may increase or decrease these values.
- Changes in revenues for increased (or decreased production) – an alternative may result in an increase in the productivity of a unit. This will result in changes in revenue.
- Increase revenue from by-products – an alternative may generate a by-product that can be sold to a recycler or sold to another company as material. This will increase a company's revenue.

It is suggested that savings in these costs be taken into consideration first, because they have a greater impact on the project's cash flows and involve less effort to estimate reliably. The remaining elements usually have a smaller impact and should be included on an as-needed basis or to fine-tune the analysis.

A project's profitability is measured by estimating the net cash flows each operating year over the life of the project. A net cash flow is calculated by subtracting the cash outlays from the cash incomes starting at the beginning of the project (the year the project is initiated).

If a project does not have an initial investment, the project's profitability can be evaluated by whether an operating cost savings occurred or not. If such a project reduces overall operating costs, it should be implemented. For example, suppose an organization currently recycles plastics and metals. If the organization currently ships comingled plastics and metals to a recycling processor, a process change could be implemented that requires employees to separate plastics from metals before shipment. There is little to no initial investment for this example, but there will be added labour costs for separation versus the additional revenue generated by the finer sort to the processor. If the additional revenues outweigh the additional costs, the alternative should be implemented.

For projects with significant initial investments or capital costs, a more detailed profitability analysis is needed. The three standard measures of profitability are:

- Payback period
- Internal rate of return (IRR)
- Net present value (NPV)

The payback period for a project is the amount of time required to recover the initial cash outlay for the project. The formula for calculating the payback period is on a pre-tax basis in years is:

$$\text{Payback Period} = \frac{\text{Capital Investment}}{\text{Annual operating cost savings}} \tag{1}$$

For example, suppose a manufacturer installs a cardboard baler for a total cost of $65,000. If the baler is expected to save the company $20,000 per year, then the payback period is 3.25 years. Payback period is typically measured in years; however, some alternatives may have payback periods in terms of months. Many organizations use the payback period as a screening method before conducting a full financial analysis. If the alternative does not meet a predetermined threshold, the alternative is rejected. Payback periods in the range of three to four years are usually considered acceptable for low risk investments. Again, this method is recommended for quick assessments of profitability. If large capital expenditures are involved, it should be followed by a more strenuous financial analysis such at the IRR and NPV.

The internal rate of return (IRR) and net present value (NPV) are both discounted cash flow techniques for determining profitability and determining if a waste minimization alternative will improve the financial position of the company. Many organizations use these methods for ranking capital projects that are competing for funds, such as the case with the various waste minimization alternatives. Capital funding for a project can depend on the ability of the project to generate positive cash flows beyond the payback period to realize an acceptable return on investment. Both the IRR and NPV recognize the time value of money by discounting the projected future net cash flows to the present. For investments with a low level of risk, an after tax IRR of 12 to 15% if typically acceptable.

The formula for NPV is:

$$NPV = \sum_{t=0}^{N} \frac{C_t}{(1+r)^t} \tag{2}$$

Each cash inflow/outflow is discounted back to its present value (PV). Then they are summed. Therefore

Where

- t - the time of the cash flow

- N - the total time of the project
- r - the discount rate (the rate of return that could be earned on an investment in the financial markets with similar risk.)
- C_t - the net cash flow (the amount of cash) at time t (for educational purposes, C_0 is commonly placed to the left of the sum to emphasize its role as the initial investment).

The internal rate of return (IRR) is a capital budgeting metric used by firms to decide whether they should make investments. It is an indicator of the efficiency of an investment, as opposed to net present value (NPV), which indicates value or magnitude. The IRR is the annualized effective compounded return rate which can be earned on the invested capital, i.e., the yield on the investment.

A project is a good investment proposition if its IRR is greater than the rate of return that could be earned by alternate investments (investing in other projects, buying bonds, or investing the money in a bank account). Thus, the IRR should be compared to any alternative costs of capital and should include an appropriate risk premium.

Mathematically, the IRR is defined as any discount rate that results in a net present value of zero for a series of cash flows. In general, if the IRR is greater than the project's cost of capital, or hurdle rate, the project will add value for the company. The equation for IRR is:

$$NPV = \sum_{t=0}^{N} \frac{C_t}{(1+r)^t} = 0 \qquad (3)$$

Most spreadsheet programs typically have the ability to automatically calculate IRR and NPV form a series of cash flows. Following is an example applying these financial evaluation concepts. For example, the baler case study discussed previously had an initial cost of \$65,000 and \$20,000 in annual savings. Additionally, the assumed baler life span was 10 years and an organization minimum attractive rate of return (MARR) was 15%. The MARR is the is the minimum return on a project that a manager is willing to accept before starting a project, given its risk and the opportunity cost of foregoing other projects. The following cash flows, IRR, and NPV result:

Year	Cash Flow
0	\$(65,000)
1	\$20,000
2	\$20,000
3	\$20,000
4	\$20,000
5	\$20,000
6	\$20,000
7	\$20,000
8	\$20,000
9	\$20,000
10	\$20,000
IRR	28.2%
NPV	\$30,761

Table 1. Net present value analysis

As shown in the last two rows of Table 1, the IRR is 28.2% and the NPV is nearly $31,000 at a MARR of 15%. The fact that the IRR is greater than the 15% MARR and the fact that the NPV is positive indicates that the project is a good financial decision.

3.3 Sustainability and organisational culture feasibility

Waste minimization alternatives should also be evaluated based on sustainability and the cultural fit within the organization. Sustainability is defined as an organization's investment in a system of life, projected to be viable on an ongoing basis that provides quality of life for all individuals and preserves natural ecosystems. Sustainability in its simplest form describes a characteristic of a process that can be maintained at a certain level indefinitely. The term, in its environmental usage, refers to the potential longevity of vital human ecological support systems, such as the planet's climatic system, systems of agriculture, industry, forestry, fisheries, and the systems on which they depend. In other words, the waste minimization alternatives should be evaluated based on how well they meet this definition, such that the alternative can be sustained without large amounts of effort or additional resources and continue to protect the environment. Often, this will be related to the culture of the organization. Criteria commonly used to evaluate the sustainability of an alternative include:

- Dealing transparently and systemically with risk, uncertainty and irreversibility
- Ensuring appropriate valuation, appreciation and restoration of nature
- Integration of environmental, social, human and economic goals in policies and activities
- Equal opportunity and community participation/Sustainable community
- Conservation of biodiversity and ecological integrity
- Ensuring inter-generational equity
- Recognizing the global integration of localities
- A commitment to best practice
- No net loss of human capital or natural capital
- The principle of continuous improvement
- The need for good governance

When an alternative involves working with a recycler or commodity broker there are several key questions to ask potential candidates to determine the best fit for the organization. Those questions include:

- What types of materials does the company accept and how must they be prepared?
- What contract terms does the buyer require?
- Who provides the transportation?
- What is the schedule of collections?
- What are the maximum allowable contaminant levels and what is the procedure for dealing with rejected loads?
- Are there minimum quantity requirements?
- Where will be recyclable material be weighed?
- Who will provide containers for recyclables?
- Can "escape clauses" be included in the contract?
- Be sure to check references.

In a similar way, when working with equipment vendors, there a several key questions to consider:

- What is the total cost of the equipment including freight and installation?
- What are the building requirements and specifications for the equipment (compressed air, electricity, space, minimum door widths)?
- Does a service contract included in the purchase price or is there an additional charge?
- Do you offer training to the employees, engineers, and maintenance employees that will be working with the equipment, if so, is there a charge?
- What is the process if the equipment malfunctions and the company needs support, is there a representative available 24 hours per day? What is the charge for these visits?
- Do you offer an acceptance test process to ensure that the equipment operates within the promised specifications (capacity and cycle time)?
- What is the required installation time and must production be shut down?

4. Case study

In 2008, the Lucas County Solid Waste Management District (District) located in Ohio, USA, considered the purchase of a material recovery facility (MRF) to sort and sell nearly 10,000 tons recyclable materials that were collected per year from its municipal recycling programs. This section analyzes the economic and operational feasibility of the MRF as an option for processing recyclable materials and may serve as an example for other local governments considering the implementation of such a system. A strong emphasis is placed on economic efficiencies and a sensitivity analysis is also discussed. A break-even analysis is discussed to determine the degree by which the existing conditions would need to change in order to allow such a facility to become feasible (or infeasible).

Based on a literature review of previous research conducted in this field, three relevant articles were found. The first was published in 1995 and is titled "The development of material recovery facilities in the United States: status and cost structure analysis" (Chang and Wang, 1995). This article examined a fast track MRF development in the U.S. and the related operating and cost structures. The purpose of the paper was to create solid waste management strategies and to aid in future investment forecasting or policy decisions. The second paper was published in 2005 and is titled "Sustainable pattern analysis of a publicly owned material recovery facility in a fast-growing urban setting under uncertainty" (Daliva and Chang, 2005). This research applied grey integer programming techniques to screen optimal shipping patterns and the outcome was an ideal MRF location and capacity design. The final paper was a report published in 1994 by the Pennsylvania Department of Environmental Protection and is titled "Lycoming County Material Recovery Facility Evaluation" (Beck, 2004). This research evaluated the operational efficiency and cost/revenue of a Lycoming County MRF. The paper also identified methods that the facility, and others like it, could be made more financially sustainable over the long term.

4.1 Methodology

The methodology used to conduct this research was based on the principles outlined in the third edition of "Facilities Planning" (Tompkins, et. al., 2003). This book provides an industrial engineering basis for defining facility requirements, identifying equipment needs, developing layouts, and implementing facility plans. This research examined the hypothesis that a county owned MRF could be cost justified and financially advantageous versus the current system of outsourcing in Lucas County, Ohio. The assumptions for this case study included:

- The useful life of the MRF is 20 years (2007 to 2027)
- A minimum attractive rate of return (MARR) of 15% was fixed over the 20 year project life for financial decisions
- Recycling levels would increase at annual rates of 5% for fiber, 3% for plastics, 2% for glass, and remain constant for metals over the 20 year project life.
- Recycling commodity prices would remain increase at a rate of 2.5% the 20 year project life.
- Utility costs would increase at a rate of 2.5% per year over the 20 year project life (inflation).
- Labour and benefit costs would increase at a rate of 3.5% per year over the 20 year project life (inflation).

The first phase of the analysis process involved estimating the current recycling levels in terms of materials compositions and volumes (annual tonnages). These data were collected from District records from the 2007 fiscal year and included operating cost and revenue data. Once combined, this information provided a complete baseline of the operations of the current system utilizing the outsourced processes. This baseline was used to compare the cost structure of acquiring a county owned and operated MRF. The baseline data provided annualized costs and revenues associated with the existing drop-off recycling program, specifically:

- Revenue paid from third party processors for recyclable materials
- Third party processing fees
- Labour costs
- Administrative costs
- Vehicle costs (fuel, maintenance, repair)
- Drop off container and material costs

The second phase involved indentifying potential MRF sites. A local business realtor was contacted for assistance. Upon the identification of the optimal MRF site, a complete annual cost and revenue projection was conducted to operate the MRF over a 20 year period. This analysis included the following annualized costs and revenues:

- Revenue paid from third party recycling material commodity brokers
- Building purchase cost (including realtor fees)
- Building modification and renovation costs
- Equipment and inspection/repair costs
- Labour costs (including driver and processors)
- Administrative costs
- Utility costs
- Vehicle costs (fuel, maintenance, repair)
- Drop off container and material costs

This financial projection of the proposed MRF was compared with the current system baseline. In essence, the analysis answered the question whether the additional revenue earned from the sale of the processed recyclable materials outweighed the additional capital and operating costs over the projected 20 year life of the project at a 15% minimum attractive rate of return. To accomplish this analysis, a net present worth (NPW) was conducted. This method not only allows the selection of a single project based on the NPW

value and in the case of this case study, the existing system of outsourcing versus purchasing and operating a county owned MRF.
To find the NPW of a project an interest rate is needed to discount the future cash flows. The most appropriate value to use for this interest rate is the rate of return that one can obtain from investing the money elsewhere. Alternatively, it may be the rate that an organization will be charged if it had to borrow the money. The selection of this rate is a policy decision by organizational management and is usually based on market conditions.
To begin this process, the District determined the net cash flow in each period over the service life of the project. Considering the MARR, each of these net cash flows was discounted back to the present time (year zero at the start of the project). The magnitude of NPW determines whether the project is accepted or rejected. If NPW is positive, the decision is to accept the project. If it is negative, then the investment is not worthwhile economically. If it is zero, then the project does not make a difference economically.
It is also possible to conduct a break-even interest rate analysis by varying the value of the interest rate while computing the NPW of a project. The break-even interest rate is the rate at which NPW is zero. The break-even interest rate is also known as the internal rate of return (IRR).

4.2 Overview of the current recycling process

Recycling services provided by the District to the local community are accomplished via a drop-off program. In Lucas County, the District collects two recycling streams from over 60 drop off sites throughout the community. These two material streams are commingled paper products and commingled containers. The drop-off sites are located at grocery stores, schools, metro parks, township offices, and large apartment complexes. Each drop off site has at least two five-cubic yard dumpsters, one for each recycling stream. At high volume sites, multiple containers are utilized for the two recycling streams. Below is a summary of the total tons of each waste collected in 2006 at the drop-off sites:

- 4,368 tons of ONP and MOP
- 2,912 tons of OCC
- 1,493 tons of glass bottles
- 677 tons of plastic bottles
- 235 tons of steel cans
- 70 tons of aluminium cans

4.3 Current system costs and revenue

Under the current system the District's drop-off program was operating at a $425,462 loss per year considering revenue minus expenses. The loss is offset by additional revenue generated by the District. The additional revenue is primarily generated from a $3 per ton surcharge on all solid waste generated in Lucas County. This surcharge is collected by the landfills that serve Lucas County and amounts to approximately $1.5 million per year.
Under the current contract the District has entered with a third party processor, the District generates the following revenue per ton of material (please note the District is paid based on commingled materials that require additional sorts):

- $37.08 per ton of commingled fiber (OCC, MOP, ONP)
- $23.35 per ton of commingled containers (aluminium/steel can and plastic)

Per year, the District generates $327,734 from the sale of recyclables to the third party processor. This revenue is offset by the following annual costs:

- $350,196 for truck diesel fuel costs
- $5,500 for annual maintenance costs
- $7,500 for drop-off site container costs and maintenance
- $240,000 for trucdk driver salaries and benefits for the four drivers employed by the District (one drive is a team leader that operates a vehicle as needed)
- $150,000 for administrative costs which include the Solid Waste District Manager's and administrative assistant's salary and benefits in addition to supply costs.

4.4 Proposed system costs and revenue

Under the proposed system the District's drop-off program will operate at an $189,327 loss per year considering revenue minus expenses. The revenue generated from the sale of the sorted recyclable materials was calculated using the current values of the Chicago material prices listed below (current as of 2/2008):

- Mixed office paper - $82/ baled ton
- White ledger - $102/baled ton
- Newspaper - $55/baled ton
- Cardboard - $110/baled ton
- Aluminum cans - $180/crushed and baled ton
- Steel cans - $180/crushed and baled ton
- Plastic bottles - $180/crushed and baled ton
- Glass bottles - $25/ton

Based on the forecasted volumes and commodity prices, the District will generate $844,197 annually from the sale of the recyclable materials to commodity brokers. From an expense standpoint, the new system will require additional money to operate and to maintain the MRF, specifically, the cost of the building, labour costs, utility, costs, maintenance costs, and management/administrative costs. The cost of the building will be addressed in the comparison and justification portion of this chapter. Specifically, the costs for the proposed system are:

- $365,100 for truck diesel fuel costs (this is up slightly from the current system due to the location of the proposed MRF and the additional required miles for the trucks to deposit material there)
- $5,500 for annual maintenance costs (no change from the current system)
- $7,500 for drop-off site container costs and maintenance (no change from the current system)
- $240,000 for truck driver salaries and benefits for the four drivers employed by the District (no change from the current system)
- $186,400 in labour costs for employees to operate the MRF (these were discussed in the previous section)
- $190,000 for administrative costs which include the Solid Waste District Manager's and administrative assistant's salary and benefits in addition to supply costs (the proposed system includes $40,000 for a District employee to supervise the MRF)
- $39,024 in utility and building maintenance costs for the MRF

The utility and building maintenance costs were estimated from the current costs of the proposed site as determined from existing records.

4.5 Financial comparison and analysis

To complete the financial analysis the Full Cost Accounting for Municipal Solid Waste, published the US Environmental Protection Agency, was used as a guide (US EPA, 2006). The proposed system will result in an annual cost savings of $236,135 versus the existing system of outsourcing. This was calculated by taking the projected annual net revenue (cost) of the proposed system minus the annual net revenue (cost) of the current system. Both system will result in a net cost for the District, (-$189,327 for the proposed system minus -$425,462 of the current system). The initial investment for the proposed system, which includes the cost of the building and renovations, is $973,050. The breakdown for this amount is $900,000 for the building and equipment and an additional $73,050 to refurbish the building and equipment. The $73,050 is the total amount provided by contactors based on inspection of the building and equipment. The payback period for the proposed system is 4.12 years (or four years and 1.5 months) and the internal rate of return for the first five years of operation is 6.8% and 20.5% for the first 10 years of operation. Working with the Lucas County Commissioners a $1,000,000 bond at 6% interest will be established with a 20 year payback period to acquire the fund for the initial investment of $973,050.

4.6 Breakeven and sensitivity analysis

From a financially standpoint, the proposed system has a payback period 4.12 years and an internal rate of return of 20.5% over 10 years based on the market assumptions stated earlier. A critical concern involves analyzing changes to these assumptions and the impact to the decision to implement. The breakeven point and a sensitivity analysis of the proposed system based on changes in market conditions will answer address this concern. From a breakeven standpoint, two market changes were analyzed:

- The lowest level that the amounts of material recycled (in tons) by the District could fall and still achieve a 10 year IRR of 6.5%
- The lowest level that the dollar values of the waste commodities could fall and still achieve a 10 year IRR of 6.5%

The breakeven point for the amount of materials collected by the District and the dollar values for the waste commodities was analyzed. An analysis of the data indicated that the amount of materials collected by the District could drop by 13% or 1,300 tons from the estimate to achieve an IRR of 6.5%. This would amount to an $110,000 reduction in revenue per year for the District. On average, the amount of materials collected by the District has increase by 3% to 5%, so this is not a large concern. Similarly, the dollar values provided by the commodity brokers based on the market rates could drop and average of 13% for each material type from the current conditions to achieve an IRR of 6.5%. This would also amount to an $110,000 reduction in revenue per year for the District.

A sensitivity analysis was conducted to determine which variables would have the largest impact on the revenue target, hence meeting the IRR, if they were reduced. To accomplish this, each variable was reduced by 5% while all other variables were held constant and the percent change in revenue was measured. The variables analyzed were:

- Amounts of materials collected (measured in tons)
- Dollar value per ton of recycling material

From this analysis OCC amounts and their price were most sensitive to changes and therefore have the largest impact on total revenue and IRR. A 5% reduction in the amount collected annually or the dollar value per ton of OCC reduced the total revenue by 2%.

Likewise, a 5% reduction in ONP reduced total revenue and IRR by 1%. All other variables did not indicate a high level of sensitivity.

4.7 Conclusions

This case study demonstrated the process for municipalities to economically justify the purchase and operation of a government owned MRF. Key findings from this research revolve around a case study from the 2008 purchase of a government owned MRF in Ohio, USA. The key findings were demonstrated through a complete financial analysis. Specifically, the financial analysis indicated that the municipality would achieve a payback period of approximately four years, and a ten year internal rate of return of 20.5%. The consequences of these findings, stemming from the economic and operational justification, led to the actual purchase of the MRF site and subsequent operation in 2008 through early 2011. This research may serve as an example or model for other local governments considering the implementation of such a system.

A strong emphasis was placed on economic efficiencies and a sensitivity analysis of the results to changes in the data inputs, specifically inflation, recycling levels, and recycling commodity market shifts. A break even analysis of the data indicates that the amount of materials collected by the District or the commodity prices could drop by 13% ($110,000) from the estimate to achieve an IRR of 6.5%. On average, the amount of materials collected by the District has increased by 3% to 5%, so this is not a large concern. The sensitivity analysis indicated that OCC amounts and price are most sensitive to changes and therefore have the largest impact on total revenue and IRR. A 5% reduction in the amount collected annually or the dollar value per ton of OCC reduces total revenue by 2%. All other variables did not indicate level a high level of sensitivity.

Reservations of limitations of this research include:

- Location and the cost of business in various geographical areas
- Inflation
- Recycling commodity market shifts
- Competition

This research and MRF analysis was conducted in the Midwest, which has a relatively lower business and real estate costs versus the East or West Cost. Conducting a similar study in these areas may not be economically justified based on these higher costs. Major changes in inflation (labour and operating costs) or commodity market shifts may alter the economics of the ten year cost structure. Finally, unforeseen competition arising in the area could reduce material collection amounts, hence reducing revenues. This competition could present itself as a new private sector recycling collector/processor or as modified fee structures from existing companies. The likelihood of these events over the ten year time frame is relatively low due to these companies current cost structures and taxation rates.

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15

Waste Management at the Construction Site

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1. Introduction

Construction and demolition (C&D) debris is produced during the construction, rehabilitation, and demolition of buildings, roads, and other structures (Clark, Jambeck, and Townsend; 2006). According to the U.S. Environmental Protection Agency (2003), C&D debris amounts to 170 million tons per year, or 40 percent of the solid waste stream in the U.S. While efforts to reduce this through reduction, recycling, reuse, or rebuying continue to expand through government mandates, green building incentives, and education, much work remains to be done. This chapter will begin with a history of C&D debris management and will cover state and local government regulations that pertain to C&D debris. Impacts on this matter from green building programs will be described. Issues that pertain to landfills, including C&D landfills, will be evaluated, along with concerns that relate to specific materials. The chapter will conclude with a discussion of lessons learned to date and recommendations for improved progress.

2. Background

Waste generation by human societies is not new. "Since humans have inhabited the earth they have generated, produced, manufactured, excreted, secreted, discarded, and otherwise disposed of all manner of wastes" (Meliosi,1981, p.1). Working to devise methods for dealing with their societies' wastes has occupied humans since the beginning of civilization and the creation of cities. The ancient city of Athens, Greece had a regulation that required that waste be dumped at least a mile away from city limits; and ancient Rome had sanitation crews in 200 AD (Trash Timeline, 1998).

What is new is the amount of waste produced by human societies, especially industrial societies. Of course part of this is driven by the rise in human population. More people will create more waste. But the amount of waste created has soared since the industrial revolution and development of a culture and global economy driven by consumption.

The development of formal management strategies for the collection and disposal of solid waste in the United Sates has occurred primarily within the last 110 to 120 years. Early America had a relatively small population that was widely dispersed on the land and relied primarily on an agrarian-based economy. Few waste materials were produced, and every possible use was sought for materials before resorting to discarding anything.

As Susan Strasser notes in her book *Waste and Want: A Social History of Trash* (1999):

"most Americans produced little trash before the 20th century. Packaged goods were becoming popular as the century began, but merchants continued to sell most food, hardware and cleaning products in bulk. Their customers practiced habits of reuse that had prevailed in agricultural communities here and abroad. Women boiled food scraps into soup or fed them to domestic animals; chickens, especially, would eat almost anything and return the favor with eggs. Durable items were passed on to people of other classes or generations, or stored in attics or basements for later use. Objects of no use to adults became playthings for children. Broken or worn-out things could be brought back to their makers, fixed by somebody handy, or taken to people who specialized in repairs. And items beyond repair might be dismantled, their parts reused or sold to junk men who sold them to manufacturers"(p.12).

While the method Strasser describes above may have worked in the sparsely populated country side, it was not perfectly suited to cities. A brief review of references to the filth of American cities of the mid 19th century made in the historical record illustrates this. For example, in Washington D.C residents in 1860 discarded garbage and chamber pots into streets and back alleys. Pigs roamed free and ate the filth, and slaughter houses emitted putrid smells. Rats and cockroaches were common in most buildings in the city, including the White House (Trash Timeline, 1998).

It was not until the late 19th century that a concerted effort started to appear in the nation's cities to clean up streets and devise some formal strategies for managing the increasing amounts of waste. Prior to that period cities typically made due with an informal network of small firms and legions of the poor who worked to collect wastes (McGowan, 1995). Citizens seldom paid to have wastes hauled away, but instead placed them at curbsides where individuals from this informal network would go through them and remove anything considered to have residual value. Items deemed to have no value were often left in the streets or tossed into alleys to rot. Milwaukee, Wisconsin provides a specific example:

"Until 1875, hogs and 'swill children'- usually immigrant youngsters trying to supplement the family income - collected whatever kitchen refuse Milwaukeeans produced. Obviously unequal to the task of collecting the wastes of an entire city these 'little garbage gatherers' left the backyards and alleys reeking with filth, smelling to heaven" (Leavitt, 1980, p. 434).

While most cities of the 19th century had no formal means of collecting and managing solid wastes, many had dumps. But they were basically open pits in the ground. Swamps were also often used as dumping grounds. Melosi (1981) cites a description given by Reverend Hugh Miller Thompson in 1879 that described a dump in New Orleans this way:

"Thither were brought the dead dogs and cats, the kitchen garbage and the like, and duly dumped. This festering, rotten mess was picked over by rag pickers and wallowed over by pigs, pigs and humans contesting for a living in it, and as the heaps increased, the odors increased also, and the mass lay corrupting under a tropical sun, dispersing pestilential fumes where the wind carried them" (p.545).

But as the industrial revolution in the United States progressed, and with the ensuing development and soaring population growth in cities across the country, cities were forced to seek more formal methods for managing wastes. In New York City in 1880 scavengers removed 15,000 horse carcasses from the streets (Trash Timeline, 1998). It was not just horse carcasses that created a problem on city streets. Engineers of that era estimated there were 26,000 horses in Brooklyn that produced 200 tons of manure and urine each day (Melosi, 1981). Most of that was deposited and left in the streets. In 1892, Milwaukee, Wisconsin citizens were in an uproar because the city's drinking water supply, drawn from Lake Michigan, had become polluted by trash and waste being dumped into the lake (Leavitt, 1980).

Driven by the waste problems illustrated by the above example, American cities began to set up trash collection programs to deal with wastes generated by their citizens. In 1880 43% of U.S. cities had a municipal program or paid private firms to collect trash. By 1900 this had increased to 65% of cities (Melosi 1981). However, there were seldom regulations on how the waste would be disposed. Many times private haulers removed any items with residual value while collecting wastes and dumped everything else in the nearest vacant lot or body of water.

As waste generation rates continued to grow and citizens complained about filthy streets and polluted water supplies, municipalities were forced to begin devising disposal methods to end these problems. The spread of disease and resulting large death toll in urban areas also spurred action. Medical thinking for much of the 19th century relied on the filth theory of disease to explain the cause of epidemics. During this time period, "most physicians believed that rotting organic wastes in crowded urban areas produced a miasmatic atmosphere conducive to the spread of diseases such as cholera, yellow fever, diphtheria, and typhoid fever" (Leavitt, 1980, p.461). This theory, even though incorrect, helped create a health justification for garbage reform (Leavitt, 1980). This is also why one of the most preferred methods of garbage and trash disposal at the turn of the century was incineration. Burning garbage and trash would sanitize it before it was hauled to a dump (McGowan, 1995). Incineration also reduced the amount of material that needed to be dumped.

Between 1900 and 1918 a national movement arose to create municipal refuse departments and bring "professional engineering and management know-how to the garbage business" (McGowan, 1995, p.155). A man named George Warring is often cited as one of the first to implement this idea in a major city. An engineer with a military background, he was appointed Sanitary Commissioner in New York City in 1894. Warring had earned a national reputation for his work in designing a modern sewage system in Memphis, Tennessee. He had been sent to Memphis by the National Board of Health after a yellow fever and cholera epidemic killed more than 10,000 people. When he came to New York he set about cleaning up the city streets and designing and building facilities to handle the city's collected garbage and trash (Melosis, p.56).

Warring had a waste recovery facility built. It consisted of a conveyor belt where immigrant laborers sorted through trash for any items of value as it passed by. The conveyor belt was powered by steam created with heat from burning trash (McGowan citing Sicular, 1984). Reduction and incineration were the preferred disposal solutions for much of the country at the beginning of the 20th century. Even those municipalities that continued land dumping saw that only as a temporary solution until they could afford to construct sorting facilities and incinerators such as Warring had built in New York (Melosi, 1981).

As a method to assist sorting at recovery facilities, many cities required their citizens to sort and separate trash before placing at the curb for collection. Spielman (2007) provides an example of one municipality's"card of instruction for householders." Residents were required to use three receptacles when putting waste materials out for collection. One was to be used for ashes. However, sawdust, floor and street sweepings, broken glass and crockery, tin cans, oyster and clam shells were also to be placed in the ash receptacle. The second receptacle was to be used for garbage. This was defined to be kitchen or table waste, vegetables, meats, fish, bones or fat. The third category was rubbish bundles. This included bottles, paper, pasteboard, rags, mattresses, old clothes, old shoes, leather and leather scrap, carpets, tobacco stems, straw, and excelsior.

Many of these advancements were abandoned with the reduction of public funding resulting from the Great Depression. Cities were forced to reconsider how to collect wastes and continue the operation of sorting facilities and incinerators. Collecting separated wastes and running sorting facilities were expensive operations. But incinerating mixed wastes made that process much more costly. The moisture content of comingled waste made it less efficient to burn (McGowan, 1995). Sanitary engineers conducted calculations to compare the cost of burning trash with burying it. These calculations showed that the cost for burning waste was two dollars per ton, while the cost of burying it just $0.29 per ton (McGowan 1995, citing Thresher, 1939). It was not long before cities quickly abandoned waste recovery and incinerator facilities and moved to the widespread practice of using dumps. Ironically, New York City also led the way in abandoning the efforts of reformers like Warring, and reinstituted the method of dumping trash on the land. William Carey, the head of New York City Sanitary Department at that time, developed dumps throughout New York's five Boroughs (McGowan, 1995). Waste was no longer viewed as a source of materials, but instead seen as "an expensive nuisance that could not be ignored" (McGowan, 1995, p. 160).

In 1931 Fresno, Jean Vincenze, the newly elected Public Works manager, immediately canceled the city's incineration contract and began what he called the sanitary fill (McGowan, 1995). Vincenze's "sanitary landfills" were nothing like current day, lined sanitary landfills. Sanitary landfills of this era used a layering process. A layer of garbage 12 inches deep would be spread over the fill area and then covered with ashes or some sort of non-putrescent rubbish. This layering process continued until the area was completely filled (Melosi, 1981).

The cost for disposing the city's trash dropped dramatically as Fresno's public works department perfected the work of collecting, transporting, and covering each day's garbage. This allowed the public works department to both expand the number of residents served with trash collection services and reduce the costs for providing this service. As McGowan(1995) notes, "the low cost and simplicity of landfill operation allowed officials of waste management firms (public and private) to concentrate their efforts on cutting costs in the labor intensive area of collection and transportation" (p.161).

2.1 C&D debris in U.S. history

A brief search of historical literature reveals little information on construction and demolition debris or how it was handled in the 19th or early 20th century. This is not surprising, since even in 1993 construction and demolition waste was seldom recorded separately from municipal solid waste (Cosper *et al.*, 1993).

Even though the country was developing at a rapid pace in the late 19th century and much new construction was undersay, a significant amount of demolition likely resulted from this development. In the October 10, 1937 issue of the *New York Times*, a story reported that "in the year 1936 there were *demolished* in the City of New York more than 10,000 dwelling units in old-law tenements and an equal number will have been *demolished* in 1937. " (Post, 1937).

Construction and demolition debris in the United States would have consisted of relatively few types of materials in the 19th and early 20th century. For example, in Philadelphia during 1950 dozens of 18th and 19th century buildings were demolished to create open space for Independence National Historical Park. During archeological work done in the park in 2000, much of the construction debris from these demolished buildings was uncovered. It was composed of wood, stone, mortar, brick, plaster, and cement (Digging in the Archives, 2010). This archival post also notes that a portion of the demolition rubble was disposed of by burying it on-site. Evidence is also cited that much of the building rubble was transported by rail to Lancaster County, Pennsylvania, where it was used to fill in a lake (Digging in the Archives, 2010).

Dumping wastes into open dumps was the most common disposal method from the period of the Great Depression until well into the 1970s. In 1972 an EPA Administrator estimated that more than 14,000 municipalities across the country relied on open dumps for waste disposal (Melosi, 1981). None of these municipalities implemented even the most basic landfill technology and attempt to layer wastes or cover each day's accumulation of trash with fill. Many of these dumps were located in wetland areas, known more commonly before the environmental movement as swamps. Abandoned gravel beds, ravines, and gullies in the landscape were also commonly used. Dumps controlled by well-managed municipalities would cover each day's accumulation of dumped waste and garbage with clean fill as a method to reduce odors and limit vermin's access to food wastes. But most dumps were merely piles of waste open to the environment. And even the best-managed landfills had no linings to protect ground water or even surface water runoff from leachate. This method of disposal continued to be the most widely used across the country until the creation of the Environmental Protection Agency and its development of strict criteria for the construction and maintenance of sanitary landfills.

The Resource Conservation and Recovery Act of 1976 (RCRA) forced the closure of open dumps across the country and developed regulations that dictated minimum standards for the construction and maintenance of sanitary landfills (Trash Timeline, 1998). Current day sanitary landfills require a liner system of compacted clay or high density polyethylene. A leachate collection system is also required to collect this liquid from the bottom of the reservoir created by the liner. Methane gas collection wells are also required. Waste is placed over the liner and leachate collection system and then covered at the end of each day with six inches of soil or an alternative daily cover (NSWMA, 2008). In some cases, inert types of construction and demolition materials are used as a daily cover material.

The closure of these dumps across the country and the expense of constructing engineered sanitary landfills significantly increased disposal costs of municipal solid waste. The increased cost of disposal began to make recycling of materials an economically viable option. In fact, recovery of materials from the waste stream did grow. It went from very small amounts to about 30% by 1995 (Spiegelman and Sheehan, 2005).

As a further method to reduce the demand for landfill space, some municipalities began to limit, and in some cases ban, construction and demolition materials from their landfills as a method to conserve landfill space. C&D materials typically do not contain putrescible wastes that sanitary landfills are designed for. In addition, many materials in C&D waste can be recovered and recycled. But even as late as 1996 only 20-30% of C&D debris was recovered for reuse or recycling. The majority of the remaining material was land-filled (U.S. EPA, 1998).

In 2003 the United States Environmental Protection Agency estimated that construction and demolition debris totaled approximately 170 million tons (U.S. EPA, 2003). This amount is broken down as follows:

- **Construction: 15 Million Tons (9% of the total).** This refers to waste materials generated during initial construction.
- **Renovations: 71million tons (42% of the total). This includes** remodeling, replacements, additions, includes wastes from adding new materials and removing old materials.
- **Building Demolition: 84 million tons (49% of the total).**

Of these amounts, the following breakdown is made:

- Residential construction: 6%
- Non-residential construction: 3%
- Residential renovation: 22%
- Non-residential renovation: 19%
- Residential demolition: 11%
- Non-residential demolition: 39%

Figure 1 displays these figures graphically.

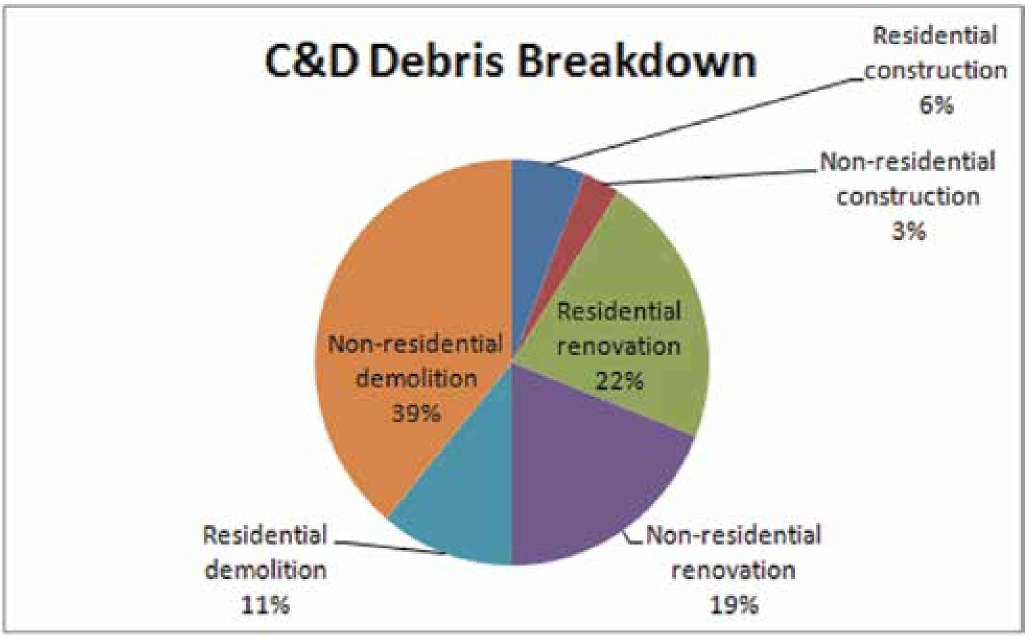

Fig. 1. C&D debris breakdown in the United States.

The EPA roughly estimates that 48% of C&D materials were recovered in 2003, which is 23% higher than the recovery estimate of 1997 (U.S. EPA, 2003). The agency also estimated that while much of the non-recovered C&D materials went to specifically designated C&D landfills, a significant amount also went to municipal solid waste landfills or incinerators. However, the amount of C&D waste co-mingled with municipal solid waste is not known (U.S. EPA, 2003).

2.2 Conclusion

Sustainability means that a community or society can continue to do what it is doing forever. But current rates of raw material inputs and energy consumption required to construct, maintain, and then dispose of buildings in the United States is certainly not sustainable for any extended period of time. And the widespread practice of simply burying construction and demolition materials instead of using those materials to reduce the amounts of raw materials extracted from the environment is a strategy that cannot be sustained indefinitely. In a world with an expanding global economy and the increasing demand for material resources, we must end the linear process currently used for material acquisition and use. We must find ways to imitate natural systems where there is no such thing as waste material, so that materials are constantly recycled and serve as inputs to the human economy or nourishment to the eco-system.

3. Federal regulations and C&D debris

While C&D debris is not explicitly regulated at the federal level in the U.S., the disposal of solid and hazardous waste is covered by the Resource Conservation and Recovery Act (RCRA) of 1976, which amended the Solid Waste Disposal Act of 1965. RCRA set national goals for:

- Protecting human health and the environment from the potential hazards of waste disposal.
- Conserving energy and natural resources.
- Reducing the amount of waste generated.
- Ensuring that wastes are managed in an environmentally-sound manner.

(U.S. EPA, 2010a)

Through the state authorization rulemaking process, the EPA has delegated RCRA implementation responsibility to individual states. Since the enactment of RCRA, other federal statutes have been passed that affect C&D debris, including the National Emission Standards for Hazardous Air Pollutants (NESHAP), which apply to asbestos, and the Comprehensive Environmental Response Compensation and Liability Act (CERCLA), also known as the Superfund, which applies to any hazardous material in C&D debris. The Toxic Substances Control Act specifically regulates the disposal of PCB ballasts in debris generated from activities related to renovation and demolition.

The 1970 Clean Air Act Amendments established NESHAP, through which the EPA is required to identify and list harmful air pollutants (EPA, 2010b). These standards require that emissions from these pollutants be minimized to the maximum extent possible through the Maximum Achievable Control Technology (MCAT). NESHAP specifies procedures for removing and disposing of asbestos.

4. State regulations and C&D debris

From the perspective of states having the primary responsibility for C&D debris regulation, Clark *et al.* (2006) provided an extensive review of individual state activities in this regard. They found a high degree of variation among states in regulatory aspects of C&D debris. At a most basic level, states vary in how they define this waste, which affects its management. Some states separately define *construction debris* and *demolition debris.* Some include it in other definitions of waste. For example, Maryland includes C&D debris in its definition of *processed debris.* Mississippi includes C&D debris in its definitions of *rubbish* and *industrial processed debris.* Other states include C&D debris in their definitions of *dry waste* or *inert waste.*

For landfills that accept C&D debris, states also vary in their regulation. California requires that such landfills be located in areas of low seismicity. Indiana specifies characteristics of the soil lining in landfills adjacent to aquifers. Not all states require that landfills have soil liners. Those that do specify a lining system of clay or other soil that meets specific requirements. Some states require leachate collection systems and groundwater monitoring.

Clark *et al.* (2006) effectively documented the wide variation among states in their regulations concerning the disposal of C&D debris. They noted differences with respect to definitions, specifically whether states defined C&D debris as one or two categories for regulatory purposes, whether inert debris was categorized, and if other definitions applied to C&D debris. They noted which states did and did not have landfill liner requirements and which had specifications for leachate collection. Permitting issues they noted were those pertaining to financial assurance and training for operators and landfill spotters. They also reported on state regulations that are specific to C&D landfills and C&D recycling facilities, groundwater monitoring requirements, and which states were updating regulations on C&D debris.
To determine if any state regulations had changed since the Clark *et al.* (2006) study, we contacted appropriate personnel of the landfill-regulating agency in every state and asked if any regulations had changed since that paper was published.

4.1 States that have not updated regulations

States that reported no changes to their C&D debris regulations since the Clark *et al.* (2006) study are shown in Table 1.

State	Notes
Alabama	
Alaska	
Arizona	
Arkansas	
Colorado	
Delaware	
Florida	
Georgia	No changes, but the state has funded a research project to examine the feasibility of more recycling.
Hawaii	No changes to regulations, but corrections should be made to Table 1 in the Clark *et al.* (2006) study: Hawaii does have a definition for construction and demolition waste, but not for construction waste, demolition waste, or inert debris. The state does have other definitions. Hawaii does require spotters, as well as training for spotters, under C&D landfill permits. The state does have regulations covering C&D debris recycling facilities.
Idaho	
Iowa	
Kentucky	No changes have been made, but regulations are currently being revised and will be presented to the state legislature in 2012.
Louisiana	No changes have been made, but the Governor's office issued an emergency declaration on August 30, 2005 to cover the disposal of 22 million tons of debris that resulted from Hurricane Katrina. 600,000 residential structures were affected; of these, 77% were completely destroyed. Over 6,000 commercial structures were affected; of these, 67% were completely destroyed (State of Louisiana, 2005).
Maine	
Maryland	No changes have been made to regulations, but a correction should

	be made to the Clark *et al.* (2006) study to note that the state does regulate C&D landfills and C&D recycling facilities. The study also notes that the state requires a final cover over a landfill of two feet of earth within 60 days and then a cap within two years, but not exactly what the cap consists of: The cap is required to have a low-permeability layer of plastic or clay, a drainage layer, a minimum slope of 4%, and at least 18" of dirt and 6" of topsoil that is compacted and vegetatively stabilized.
Massachusetts	As of July 2011, clean gypsum wallboard will be added to asphalt pavement, brick, concrete, metal, and wood on the list of materials banned from disposal.
Michigan	No changes have been made except generic exemptions for asphalt shingles, new construction drywall, and scrap wood.
Mississippi	
Missouri	
Nevada	
New Hampshire	The discussion about capping systems should be revised to reflect specifications in the New Hampshire Code of Administrative Rules, Part 805.10.
New Jersey	
New Mexico	New Mexico implemented new general solid waste rules in August 2007. Regarding C&D debris, however, no changes have been made.
New York	
North Carolina	Senate Bill 1492 passed in 2007. It has enhanced protections applicable to sanitary landfills, which pertain to C&D debris, that are not in the rules. In particular NEW C&D landfills, of which there are none , and permitted after August 1, 2007, are required to have liners and leachate collection systems. Of particular note is the buffer requirements to parks, wildlife preserves and hunting lands.
Ohio	New rules and programs are currently being adopted, but they are in the early stages. Otherwise, no changes have been made.
Oregon	
Pennsylvania	
Rhode Island	
South Dakota	
Texas	
Utah	
Vermont	
Washington	Most of the information provided in Clark *et al.* (2006) is incorrect or confusing. Washington amended its solid waste rules in 2003, well before that paper was published. State personnel found that the wrong agency had responded to the request for information (WA Dept of Natural Resources) and the wrong regulation was referenced. The current rule no longer includes definitions of demolition or construction waste but has a definition for inert waste as well as standards for inert waste landfills. This is covered in sections 100,

	410, and 990 of WAC 173-350, Solid Waste Handling Standards. Non-inert construction and demolition waste destined for disposal must be managed in either a limited purpose landfill authorized to accept it (Section 400) or at a municipal solid waste (Subtitle D) landfill permitted and operated in accordance with WAC 173-351, *Criteria for municipal solid waste landfills.* These are the only three categories of landfill facilities in the state.
West Virginia	
Wisconsin	
Wyoming	

Table 1. States that have not changed regulations since publication of Clark *et al.* (2006).

4.2 States that have updated regulations

State	Regulation
California	We were unable to find a state official who could give a clear answer on whether regulations had changed. However, Clark *et al.* (2006) cited a definition of inert waste from the California Integrated Waste Management Board as: "Subset of solid waste that does not contain hazardous waste or soluble pollutants at concentrations in excess of applicable water quality objectives, and does not contain significant quantities of decomposable waste." Clark *et al.* (2006), p. 150. We found that the California Integrated Waste Management Board is now the Department of Resources Recycling and Recovery. This definition is provided: "Inert debris means solid waste and recyclable materials that are source separated or separated for reuse and do not contain hazardous waste…or soluble pollutants at concentrations in excess of applicable water quality." Regulations: Title 14, Natural Resources—Division 7, CIWMB. Chapter 3. Minimum Standards for Solid Waste Handling and Disposal, Section 17388. Definitions.
Connecticut	There has been one change to regulations that affect C&D debris since 2005. On May 31, 2006, the state issued the ruling "General Permit for Storage and Processing of Asphalt Roofing Shingle Waste and/or for the Storage and Distribution of Ground Asphalt Aggregate for Beneficial Use." See link: http://www.ct.gov/dep/lib/Permits and Licenses/Waste General Permits/Asphalt roofing shingles gp.pdf
Illinois	New regulations effective in 2009. See link: http://www.epa.state.il.us/land/ccdd/index.html
Indiana	Pulverizing is now banned. Material must be recognizable.
Kansas	Kansas' definition of C&D waste is written to prohibit disposal of

	chemical containers in C&D landfills even if empty. This requirement is why there is no groundwater monitoring. The regulations are available online at the following link: http://www.kdheks.gov/waste/regsstatutes/sw_laws.pdf.. The definition of C&D waste can be found in our state law at the same website at K.S.A. 65-3402(u).
Minnesota	A new rulemaking is underway to address financial assurance and siting requirements. This was initiated at the request of the legislature. The scope of the rule has narrowed to potentially affect only new facilities. The rule revisions were too unwieldy to deal with as one rulemaking and have been split into two: one to address financial assurance and the other to address siting requirements. Current rulemaking can be viewed at this link: http://www.pca.state.mn.us/index.php/waste/waste-permits-and-rules/waste-rulemaking/financial-assurance-and-siting-fasit-rulemaking.html This rulemaking reflects two legislative directives to to improve siting rules to better protect groundwater and improve financial assurance to assure that Minnesota taxpayers are protected, and puts a moratorium on siting or expanding many landfills until such rules are in place.
Montana	New rules for general waste management were issued in February, 2010. Minor changes to wording were included, but no major regulatory changes were made.
Nebraska	Random inspections of incoming loads are required to exclude regulated hazardous wastes or PCB wastes. Personnel must be trained to recognize regulated hazardous wastes and PCB waste. The effective date of Title 132 - Integrated Solid Waste Management Regulations is December 28, 2009. Agency personnel noted the following quotation from Clark *et al.* (2006): "Not only are some discrete components found in buildings hazardous wastes, but the buildings themselves may be hazardous wastes if painted or contaminated with toxic chemicals (e.g., coated with lead-based paint)." Clark *et al.* (2006), p. 144. The department's opinion on that topic is that under the hazardous waste regulations (Title 128) waste determinations are based on the waste "as generated." A demolition then would mean the waste "as generated" is the entire structure. It is not possible for a representative sample of the entire structure to fail a TCLP for metals. The entire mass of the waste versus the small amount of paint in relation to that waste effectively dilutes the results to well below any toxicity characteristic regulatory limits. This is obviously not intentional dilution so it is not affected by the LDR dilution prohibition. There is the remote possibility that a building might have been contaminated with a listed hazardous waste and, as such,

	the entire waste (the building debris) will be a listed hazardous waste under the mixture rule (Title 128, Chapter 2, Section 005.02). It would be possible to do a so-called contained-out determination of the debris if it could be suitably demonstrated the waste contained so little of the listed component that it presents no risk to human health or the environment.
North Dakota	Changes include: Minimize erosion and optimize drainage of precipitation falling on the landfill. The grade of slopes may not be less than three percent, nor more than fifteen percent, unless the applicant or permittee provides justification to show steeper slopes are stable and will not result in long-term surface soil loss in excess of two tons [1.82 metric tons] per acre per year. In no instance may slopes exceed twenty-five percent. Refer to North Dakota Century Code (NDCC), North Dakota Administrative Code (NDAC) Code 33-20-04.1-09 paragraph 4b3.
Oklahoma	Changes became effective July 11, 2010 and included the amendment of certain rules that directly affect C&D facilities. These changes include the following: 1. OAC 252:515, Subchapter 15: The exemption for C&D landfills was removed. This means that C&D landfills will be required to implement methane gas monitoring and control which includes the installation of gas probes, the submittal of an explosive gas monitoring and analysis plan to DEQ, and procedures for corrective action if explosive gas levels are exceeded. 2. OAC 252:515, Subchapter 29: The exception for C&D landfills was removed. This means that C&D landfills are required to have a waste exclusion plan (WEP). The key dates for implementing these rule changes are as follows. 1. OAC 252:515, Subchapter 15: a. An explosive gas monitoring and analysis plan (Plan), as required in OAC 252:515-15-3(a), must be submitted to the DEQ for approval no later than January 7, 2011. b.The Plan must be implemented no later than 90 days after it is approved by the DEQ. 2. OAC 252:515, Subchapter 29: a. A WEP, as required by OAC 252:515-29-2(a), must be submitted to the DEQ for approval no later than January 7, 2011.
South Carolina	New regulations went into effect in 05/2008. No major changes to requirements but some terminology changes – see http://www.scdhec.gov/environment/lwm/html/solidwaste_new_regulation.htm Regulation Code: 61-107.19.
Tennessee	All landfills are now to have groundwater monitoring. Cover frequency used to be less frequent, but is now once per week.
Virginia	Virginia Solid Waste Management Regulation, 9VAC20, Chapter 81 to be posted to the Virginia Department of Environmental Quality

	website as of March 16, 2011. Inert waste is no longer defined in the regulation, and the definition of C&D landfill has changed: "Construction/demolition/debris landfill" or "CDD landfill" means a land burial facility engineered, constructed and operated to contain and isolate construction waste, demolition waste, debris waste, split tires, and white goods, or combinations of the above solid wastes. Leachate control and monitoring are required. Gas management is required unless the operator can demonstrate that gas formation is not a concern.

Table 2. States that have changed regulations since publication of Clark *et al.* (2006).

Tables 1 and 2 demonstrate the difficulties associated with presenting a clear picture of the issue of C&D debris management across the United States. Some states have detailed definitions and management policies for C&D debris and the facilities that handle it. Specifications for landfill liners and covers vary, as do requirements for leachate management. Problematic issues related to gypsum wallboard waste are highlighted by the ban on disposal of this waste by Massachusetts. And as can be seen in the case of Washington, despite their meticulous research approach, Clark *et al.* (2006) were misled by a staff person in a state agency who thought his agency had regulatory authority over C&D debris when it did not.

Among other issues, the management of C&D debris has implications for water and air quality. A state with minimal oversight of such debris can affect the quality of air and water in adjacent states. Policy implications of this situation may include regional cooperation among states in their management of C&D debris, at a minimum. In addition, policies need to be communicated clearly so that those involved in construction, demolition, and related industries can remain in compliance in ways that do not have negative impacts on housing affordability and other issues.

5. Local municipal programs

Many local governments have instituted programs and issued regulations as a method to reduce the amount of C&D waste flowing to local landfills. Three examples of specific local programs are described below.

The city of Portland Oregon provides an example of a local municipality that has set regulations that require the general contractor of all building projects costing over $50,000 to make certain that 75% of the waste produced on the job-site be recycled. The general contractor is responsible for setting up a recycling program, including containers or storage areas separate from garbage for materials being recycled. The general contractor must complete a pre-construction recycling plan that details precisely how/where the following materials will be recycled:

- Rubble (concrete and asphalt)
- Land clearing debris
- Corrugated cardboard
- Metals
- Wood (City of Portland, Oregon, 2011).

The City of Austin, Texas provides an example of a municipality that uses a green building program to provide incentives to reduce construction wastes. The program sets minimum recycling and/or reuse levels of construction waste if buildings are to qualify for the Austin Energy Green Building designation. Waste reduction and recycling requirements set forth in program are designed to assist the city in meeting a waste reduction goal that calls for a 90% reduction in materials sent to landfill by 2040 (Austin Energy, 2010).

As part of the requirements that builders and developers must meet to obtain the Austin Energy Green Building designation, they must set aside space on the construction site for sorting and temporary storage of reusable/recyclable materials. Builders are allowed to reuse many of the waste materials on-site. For example, waste wood and cleared brush can be chipped and used for on-site landscaping purposes. Gypsum drywall scraps can be ground on site and used as a soil amendment. Concrete can be crushed and used as fill or drainage under garden beds or driveway areas. The program requires that a minimum of 50% of the waste generated by the construction project must be recycled or reused (Austin Energy, 2010).

The city of Seattle has also set very ambitious targets for reducing waste materials. The city has set a goal to reach a 70% recycling target by 2025. As a method to reduce construction waste, the city provides educational materials to contractors and developers on methods to reduce construction waste. They have an on-line checklist that describes basic steps in setting up a job-site reuse and recycling strategy. In addition, the following on-line resources are also provided: (1) A searchable data base for recycling construction and demolition waste, and (2) A recycling directory to identify what materials are easiest to recycle in the region (City of Seattle, n.d.).

6. Green building programs and C&D debris

Besides regulation, incentives exist for managing C&D debris in ways other than disposal in landfills. A number of green building programs are now in effect at the national, state, and local levels throughout the U.S. The most well-known of these is Leadership in Energy and Environmental Design (LEED), which is administered by the U.S. Green Building Council (USGBC). LEED is a program through which buildings are certified as meeting sustainability standards. LEED focuses on specific areas environmental health, including resource efficiency. Points are awarded to a development project for minimizing the amount of C&D debris that is sent to landfills. LEED is applicable to all buildings, including homes.

Since 2004 Enterprise Community Partners has administered the only national program to develop green homes for low-income families. The Green Communities Criteria established under this initiative relate to design, neighborhood fabric, resource efficiency, environmental health, and maintenance. This program features green characteristics that are found in many LEED buildings, but differs in its focus on serving low-income families. This effort also has a focus on minimization of C&D debris that is sent to landfills.

With input from several thousand stakeholders, the National Association of Home Builders (NAHB), the International Code Council (ICC), and the NAHB Research Center developed ICC-700, the National Green Building Standard. It was approved in 2009 as an American National Standard, and is the only green standard that is consistent with ICC's I-Codes. Green features covered by this standard are similar to those in use by LEED and Enterprise. ICC Codes are used as the basis of building codes in use across the United States.

The EPA Indoor airPlus program of the U.S. Environmental Protection Agency is an enhancement to the ENERGY STAR Home program. ENERGY STAR homes are certified to perform to a level of energy efficiency that is typically 20 – 30 percent higher than conventional homes. To be certified as an Indoor airPlus home, over 30 additional construction features are added to the home, including resource efficiency.

An implication of more widespread adoption of green building programs would be an increased awareness of the amount of construction debris that can be diverted from landfills. And as green buildings are planned in advance for deconstruction, less demolition debris will be produced.

7. The issue of gypsum

One issue that has posed challenges to C&D recycling is that of gypsum wallboard waste. This wallboard is comprised of gypsum with paper facing and backing. Gypsum is calcium sulfate dihydrate, a mineral that is mined from dried sea beds. It is the most common interior wall finish material used in new construction and remodeling in the United States (CalRecycle, 2007).

Gypsum board, also widely known as drywall or mistakenly as the brand name of a U.S. Gypsum Corporation product, Sheetrock®, generally makes up the largest single component in the C&D construction waste stream. A Cornell University study found that, on average, some 1,700 pounds of gypsum waste is produced per home constructed, amounting to approximately one pound per square foot of house area (Laquatra and Pierce, 2004).

The usual method of finishing drywall, the use of tape and joint compound to cover joints and screw depressions, is most efficiently done when the largest possible pieces of drywall are used to reduce the number of joints. This in turn requires cutting openings for doorways, windows, heating/air conditioning vents, electric receptacle and switch boxes, and junction boxes for light fixtures (as opposed to piecing multiple drywall sheets together to form openings). This produces the bulk of construction drywall waste.

Management of drywall waste may involve either disposal or recycling. Frequently, drywall waste is disposed of by simply dumping it in landfills. The chemical composition of the gypsum used in drywall, however, presents at least one important obstacle to disposing of such waste in this manner.

Many landfills in the United States now recover and use the methane gas produced by decomposition of buried organic waste. Sulfate-reducing bacteria, which thrive in the anaerobic conditions of landfills, produce hydrogen sulfide gas as they break down the sulfites in gypsum. Hydrogen sulfide gas has a foul odor and can make people sick. It is lethal in high concentrations. In addition, the presence of this hydrogen sulfide in methane recovered from landfills reduces the quality of the methane gas. Although technology is available to lessen the amount of hydrogen sulfide in recovered methane, the added expense of doing so prevents many landfills from accepting drywall waste.

One suggested method of drywall disposal is to cut scraps into small pieces and then place them in the uninsulated cavities of interior partitions (Yost, 1997). This technique has yet to be widely used, in part because of the additional labor required.

As an alternative to simply disposing of drywall waste, recycling technology has advanced to the stage where builders are now able to separate gypsum from other waste materials onsite to be picked up by drywall recyclers at costs comparable to those of landfill disposal.

Recycled gypsum from residential construction waste is used in the manufacturing of new drywall, the making of cement, as filler in stucco, as a precipitant to remove solids from turbid water, and as an absorbent to dewater the resulting sludge, in the treatment of waste water, and in the production of cat litter. Another disposal option is the reduction of waste gypsum to a powder, which because of its alkalinity, may be used in agriculture to increase the pH of overly-acidic soil. Some states, however, do not permit this.

Green building programs are now having an impact on drywall recycling. For example, USA Gypsum of Lancaster, Pennsylvania reports that much of the demand for their waste gypsum collection and recycling services is driven by requirements of green building programs (Weaver, 2011). One of the most widely known green building designation programs is LEED, which was developed by the United States Green Building Council to provide third-party verification that a building is designed and constructed to meet strict environmental criteria (USBC, 2008). One of the requirements of this certification program is that a certain percentage of the waste materials generated during construction, including gypsum, be recycled.

Even in relatively remote areas of northern NY, several hundred miles from USA Gypsum processing facilities in Pennsylvania, building contractors are willing to pay the additional costs for collection, transportation, and fees to accept the scrap gypsum for recycling if it is required to obtain the green certification for the building they are constructing (Weaver, 2011).

While the increased demand for gypsum waste recycling created by green building programs is a positive step toward reducing the amount of gypsum being land filled, there are several factors creating significant barriers to more widespread recycling of waste gypsum board. None of these factors is more significant than the movement of gypsum wall board manufactures to begin using synthetic gypsum as the preferred input to produce new wallboard. Synthetic gypsum is formed as a by-product of the process used to remove sulfur dioxide from exhaust flue gasses of coal-fired electric plants. Synthetic gypsum and naturally occurring gypsum ore are virtually chemically identical. Older gypsum board plants were capable of using a percentage of synthetic gypsum mixed in with pure natural ore. But newly constructed gypsum board plants have been designed to produce wall board without using any natural gypsum ore (U.S. Gypsum Association, 2008).

While these modern plants also have an increased capacity to accept ground gypsum processed from recycled gypsum wallboard scraps, it is currently not economically possible for gypsum board recycling firms such as USA Gypsum to compete with synthetic gypsum (Weaver, 2011). Wallboard manufacturers typically receive synthetic gypsum at no cost from coal fired electric plants. The only expense to the board manufacturing plant is the cost of transportation to get the synthetic gypsum from the power plant to the board manufacturing facility. In some cases gypsum wallboard manufacturers are building production facilities right next to coal fired electric plants as a method to minimize transportation costs.

The use of a waste product produced by one industry, the coal fired electric industry, as a raw material input for a different industry, the gypsum board manufacturing industry, is definitely a sign of progress toward moving from a linear system of resource consumption to that of a circular system where waste products of one firm serve as material inputs for another. But we are still left with the issue of how to divert millions of tons of gypsum wallboard scrap created by the construction industry from the nation's landfills. As noted earlier, decomposing gypsum in landfills produces foul smelling hydrogen sulfide gas that can create health and air quality issues for residents living miles away from the landfill. In addition, the presence of hydrogen sulfide gas reduces the quality of the methane gas recovered from the landfill. Because of these issues many local legislators and state environmental departments across the country are considering increasing tip fees for scrap gypsum or banning it from landfills altogether and requiring recycling of scrap gypsum (Breslin, 2010). These steps would create incentives to increase recycling of scrap gypsum generated by the building construction industry.

8. Waste avoidance

The growing complexity of issues related to C&D debris highlight the importance of adopting practices that minimize its production. Green building programs provide examples of incentives available to builders and remodelers for reducing the amount of debris sent to landfills. Ikuma *et al.* (in press) discuss *lean construction* as a means for using building materials efficiently, through adoption of modular building methods and increased use of factory-built panels. They presented case studies that showed a 64% reduction in material waste through adoption of the technique.

Another method for reducing the amount of C&D debris produced at a construction site involves the use of *advanced framing.* This term refers to a building technique that reduces the amount of lumber used in wood-framed houses in a way that does not sacrifice stability of the structure. Vertical framing members (studs) are aligned with those placed horizontally (joists) or at an angle (rafters). Supporting members over doors and windows (headers) are sized in a way that eliminates the need for excess wood that is commonly used. Corners are framed in ways that use wood more efficiently and allow for more insulation than is possible with conventional, overbuilt corners (NAHB Research Center, 2008).

9. Conclusions and implications

Human civilization has come full circle in its dealings with management of waste. Early humans were careful with things they produced. They reused them, and then repaired them when they broke. This was common practice until the Industrial Revolution, which made materials commonly available. Coal-fired equipment provided the ability for the production of large amounts of inexpensive goods, which then led to increasing amounts of waste (Waste Online, 2004). This same pattern was observed in the construction industry, which evolved from careful reuse of timbers for ships and buildings to sending demolition debris and construction waste to landfills. C&D debris was initially considered to be environmentally benign, but that perception gradually changed with a growing focus on hazardous components of this debris, including lead and other heavy metals, asbestos, arsenic, polychlorinated biphenyls, and others (Clark *et al.,* 2006).

Until the early 1990s, C&D debris was routinely sent to landfills, with little attention given to recycling or reuse options (Goldstein, 2006). While early landfills were essentially holes in the ground, modern landfills are now lined with compacted clay, high density polyethylene (HDPE), or other materials. None of these lining materials provides a layer that can be considered as impermeable indefinitely. Compacted clay can crack, synthetic fibers can leak, and landfill vents have been observed to release high levels of toxins (NEWMOA, 2010). C&D debris that has been deposited in either lined or unlined landfills is cause for concerns related to environmental contamination.

While the 170 million tons of C&D debris produced annually in the United States is managed considerably better now than it has been in the past, this chapter has shown where improvements can be made. The amount of this debris that is produced, for example, can be substantially reduced by producing less of it, through techniques known as lean construction and advanced framing. Efficiency improvements in the construction process will lead to less waste in construction materials and debris that is sent to landfills. Green building programs are expanding awareness of recycling and reuse options and may have substantial impacts on the reduction of C&D debris produced as those programs move further into the mainstream.

Government efforts at all levels – federal, state, and local – need to demonstrate a higher level of awareness of issues related to C&D debris. While the federal government has mostly left management of C&D debris to the states, the wide variation among states in this management shows that some states take on a strong management role to prevent pollution of groundwater and air, while others provide minimal oversight. And the case of Washington demonstrated that state agency personnel are not always aware of which agency has oversight responsibility. But these are issues that may be handled well at the local level, as our discussion of local government actions by Portland, Austin, and Seattle demonstrated.

Some building materials, most notably gypsum wallboard, may receive more attention in the coming years. Once regarded as a benign waste material, its role with gas production in landfills is receiving more notice, as is seen with the ban on its disposal in Massachusetts.

While regulation of C&D debris varies throughout the United States, an educational approach designed for those involved in all phases of its life cycle, from production to disposal or reuse, would be beneficial. Builders could participate more widely in green building programs; and purchasers, managers, and occupants of all types of buildings could become more aware of these programs and exert demand-side pressure for supply side participation. Those involved in all aspects of the building industry, from the creation of a structure until its demise, need to assume greater responsibility for improved management of C&D debris in the United States.

10. Acknowledgment

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16

Deconstruction Roles in the Construction and Demolition Waste Management in Portugal - From Design to Site Management

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1. Introduction

In the last few years, the impact of construction industry on the environment has been increasingly recognized and has become a key challenge for the sector. Construction sites activities in urban areas may cause damage to the environment, interfering in the day life of local residents, that frequently claim against dust, mud, noise, traffic delay, space intrusion, materials or waste deposition in public space, etc.. In a time where it can be seen quality improvements in construction process techniques, in materials innovation and in safety and healthy conditions, it is also necessary to take care of the environment and other sustainability related issues.

The number of new constructions in Portugal had a significant decrease on the last years. This is due to the fact that housing needs are already completely fulfilled - one dwelling per each two inhabitants. This is the result of a construction boom that took place during the 80s and 90s of the past century. But many of these buildings were made without a sustainable cost/benefit ratio and without reuse / recycling strategies, due to initial budget limitations and lack of knowledge.

In recent years, the implementation of Energetic Certification by Decree-Law 78/2006, from 4th of April, following the 2002/91/EC directive as well as new regulation on Buildings' construction waste management, Decree-Law 46/2008, from 12th of March, following the 2006/12/EC directive, conducted to relevant changes, especially regarding envelope walls, but also with repercussions on the interior layouts. There is a need of refurbishment that in some cases reflects both in the quality improvement of the construction, but also in the increase of the internal areas. The internal minimum areas have increased significantly in the last 50 years, and almost doubled, what made many buildings obsolete and not capable of fulfilling the contemporary needs of the households. Maybe this is the reason why the majority (66,9%) of the refurbishment building works taking place in Portugal in the last years correspond to extensions. Refurbishment works and rehabilitation without extensions correspond to 33,1% (INE, 2010[1]).

There are still many unoccupied dwellings (11%) and a lot of buildings needing refurbishment. But in fact the number of refurbishment works is not increasing, just the opposite, it has been slightly decreasing since 1996 as the Figure 1 evidences. However, the percentage of refurbishment works has increased slightly from 2008 to 2009, in 2,2%.

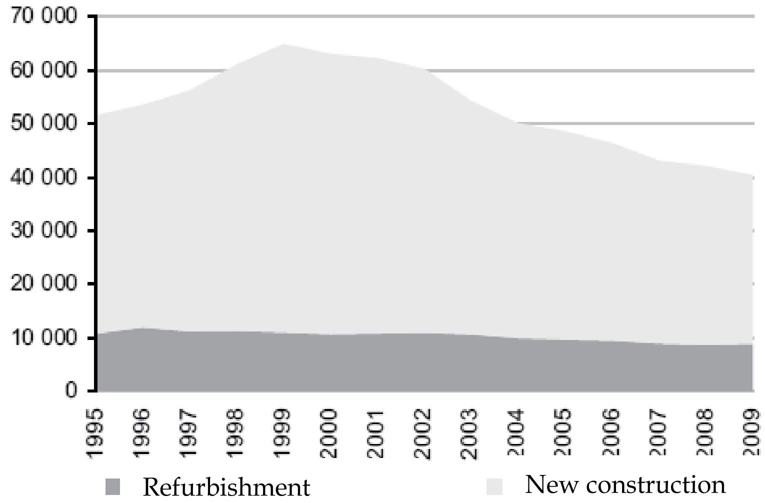

Fig. 1. Refurbished and new constructions in Portugal from 1995 to 2009. Source:(INE, 2010[1])

There is an enormous building stock in Portugal that is waiting to be refurbished. Paradoxically, very little rehabilitation takes place in Portugal — indeed. According to Euroconstruct 2008 Report in the year 2007 it was invested in refurbishment about 26% of total construction investment, whereas in other European countries it raised to about 45% (including residential, non-residential and civil engineering renovation) (Euroconstruct, 2008). The lack of interest in refurbishment underpins behaviours that limit sustainability improvement in the construction sector. The attitude is partly connected to the fact that building refurbishment involves knowledge of building materials and techniques that have been superseded. More often than not, the refurbishment of a building will stop at the preservation or restoration of the facade, disregarding the reuse of the materials inside, even though in some cases they can be recovered and employed in the new intervention. Decree-Law 46/2008 imposes since 2008 some measures in this way.

The building activity at Portuguese city centres tends to be an important waste generator because both refurbishment projects and new projects often include demolition (Couto & Couto, 2009). Surveys conducted in several countries found that the amount of waste generated by the construction and demolition activity is as high as 20–30 percent of the total waste entering landfills throughout the world and the weight of the generated demolition waste is more than twice the weight of the generated construction waste (Bossink & Brouwers, 1996). Other studies compared new construction with refurbishment, and concluded that the latter accounts with more than 80 percent of the total amount of waste produced by construction activity as a whole.

Between 2004 and 2009 Portugal generated 172 million tons of wastes mainly coming from the Transforming and the Commerce and Services Industries sector. In 2009 production decreased almost 1/4th in relation to the previous year, mainly because of the strong decrease from the Building Industry, fixing on the 24 million tons (INE, 2010[2]). Although an increase on the wastes generated by extractive industries could be seen, in result from the research and exploration of stone quarrying and mining industries, as well as from cement industries, thus a fact in direct strong relation with building activities.

Year	2004	2005	2006	2007	2008	2009
Construction sector (tons)	2 625 930	5 212 520	3 607 232	5 674 248	8 148 290	3 152 098
Total (tons)	**24 689 088**	**31 096 302**	**31 155 301**	**30 240 562**	**31 591 727**	**23 659 876**

Table 1. Wastes generated by the construction sector in Portugal between 2004 and 2009. Source: (INE, 2010[2])

The "mining" industry has shown a dynamic growth over the period under review, as evidenced by the average annual rate of around 30% recorded during this period, as documented in the following figure.

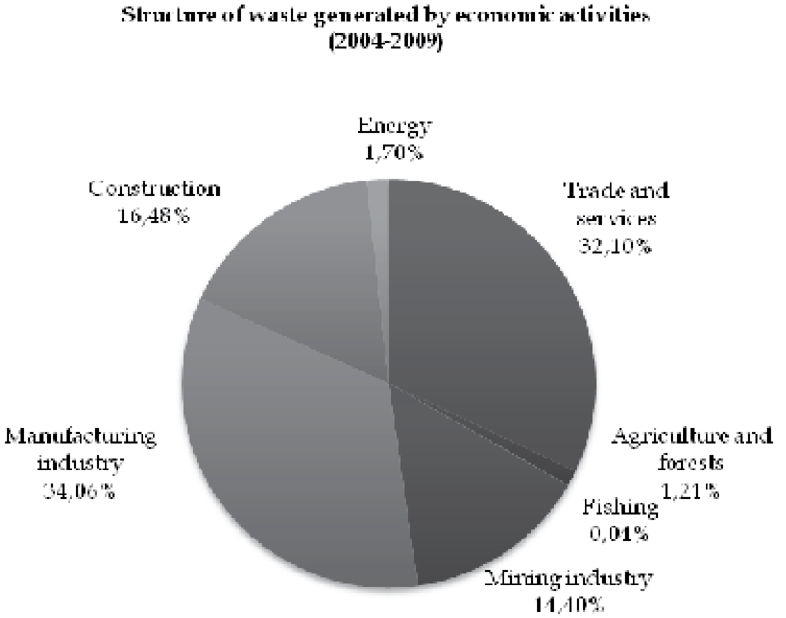

Fig. 2. Structure of waste generated by economic activities in Portugal from 2004 to 2009. Source: (INE, 2010[2])

The portion gained from the quantities of generated wastes by Gross Domestic Product (GDP), translates the efficiency level of the economy that will be as much efficient as less is the quantity of wastes per unity of generated GDP. In generic terms, the year 2009 stands as the most efficient in environmental terms, although this result is influenced by the decrease of production in general and of building sector in particular that, in relation to 2008, generated around 5 million tons less wastes. To this fact is not indifferent the implementation of the Decree-Law 46/2008 that, among other measures, preconizes the possibility of reusing soils and stones without dangerous substances, with origin on building construction, in other works, apart from the original one, as well as on the environmental refurbishment, allowing this way to avoid the waste production and simultaneously preserving the natural resources used to identical uses (INE, 2010[2]).

Construction industry rely nowadays on materials of a complex life-cycle, making use of many different raw materials and some with a high energy cost (in relation to its function),

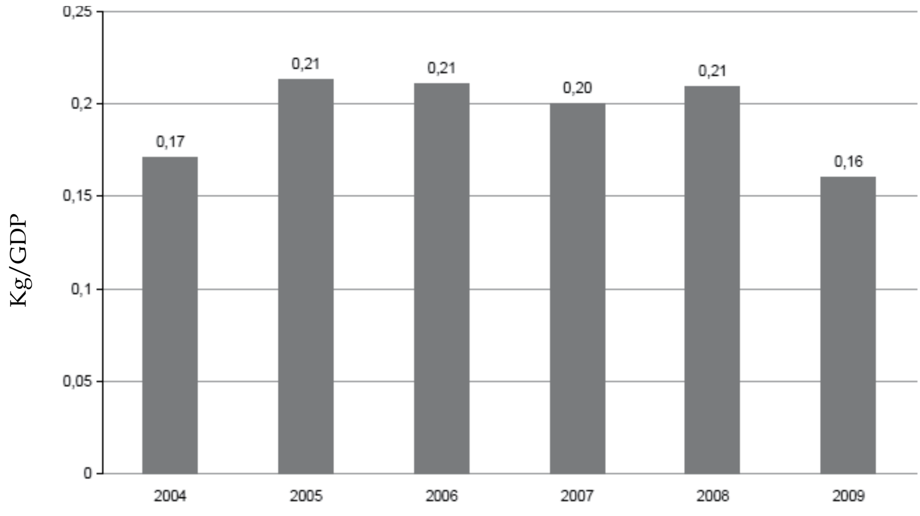

Fig. 3. Wastes generated by GDP. Source: (INE, 2010[2])

in detriment from low-energy, less transformed, recycled and preferably re-used ones. The maximum use of reused materials means reduction of environmental impacts due to the extraction of prime materials, to their transformation processes and to the work yards, with reduction of the noise, dust, wastes and the consumption of energy during the construction and a proportional reduction on loss factors and on transport energy.

Berge (2000) refers: "the amount of energy that actually goes into the production of building materials is between 6 and 20% of the total energy consumption during 50 years of use, depending on the building method, climate, etc". This is not a very relevant percentage, even if we consider the maximum, but energy cost will certainly increase in future years, and the dismantling, treatment and transport of waste materials also represents energy, especially in nowadays most common constructive system used in South European housing – concrete structure with clay hollow brick walls and pavements (Mendonca & Braganca, 2001).

Sustainability on building sector is a pluridisciplinar concept that, for its implementation, requires the cumplicity of all the involved agents, from polititians to urbanists, that have to legislate and define the planning instruments, to projectists that have to conceive efficient buildings on the resources optimization, till construtors, that should be able to construct the building in the most reasonable way.

Sustainable approach to building construction, as well as to many other areas of industry, rely on four strategies: reuse, recycling, recovery (energy) and reducing. All those points are relatively neglected in South European buildings, and specially referring the Portuguese case and, in spite of studies being made, implementation suffers a strong resistance (Mendonca & Braganca, 2001). First point focused, reuse, is usually implemented in a very limited way. Preconception about innovative materials and construction methods leads to focus the attention just on reducing environmental impact in making traditional materials for conventional buildings.

In what respects the structure and the materials used, housing constructions in South European climates are generally heavyweight. Concrete, brick or stone are used in the exterior envelope walls and structure, in order to achieve high thermal storage capacity and structural resistance. When these materials and labor are locally available (as earth, wood or stone), their environmental cost is reduced, but the increase of the global mass of the building implies other problems, such as the increasing economical cost of an high intensive labor. Some building elements cannot be always locally made, (such as steel, concrete, glass or ceramics), and in a high density multi-storey building, the percentage of the industrial and more transformed components usually increases (Mendonca & Braganca, 2001).

2. Impact of construction industry on the environment

The Building industry is a great consumer of raw materials and energy; to whom are associated the sequent pollutant emissions, associated to extraction and production of the building materials, as well as to the use phase and eventual demolition/refurbishment. Fossil fuels burning is the most important source of pollution, associated with energy needs in the use phase as well as in the first phases of extraction, producing and transport.

To evaluate the environmental impact of a building during its life cycle, it can be considered two distinct essential components: energetic and material, that are usually associated.

The environmental impact during the construction phase constitutes a much smaller percentage in relation to the production of materials, on Portuguese present reality. This is due to the use of industrialized materials, with high specific embodied energy, as well as to a bad waste management.

A principle for future actuation should consist on a drastic reduction on the use of unprocessed raw materials. This is an important factor to be considered for the most scarce resources, but should also be considered for the most abundant.

The environmental impacts of buildings and materials do not end up in the useful life term, and can be even more significant if deconstruction strategies were not considered on the design stage. During demolition or partial dismantling, the two most significant parameters that should be considered are:

- Energy consumption and worn of equipment necessary for demolishing or dismantling, as well as hand labor;
- Transport of wastes to landfill or recycling units. The building industry in Portugal was responsible for over 8 million tons of solid wastes in 2008 (INE, 2010[2]).

The environmental impacts of buildings during its useful life can be represented through a, diagram of "inputs" and "outputs", such as the one presented on Figure 4. In the "inputs" are included energy and materials and in "outputs" pollution and wastes.

In an open cycle (linear) system, representative of the Portuguese scenario for buildings constructed nowadays and in the past decades, environmental impacts of a building correspond to the sum of inputs and outputs from all the building life cycle phases represented on Figure 4.

There are several ways to promote waste management in buildings. Part of the responsibility is in the hands of building constructor, which should act with ethic principles that should go far beyond what imposes legislation, but is also mission of the architects and engineers that design the building, to give it the maximum qualities that allow an efficient waste management. Of course it is first responsibility of politicians and technicians that assessor these, to legislate about environmental issues in building construction, in order that promoters and builders feel obliged to included these aspects as major concerns, and not only the profits (Mendonca, 2005). But, before taking any action to reduce environmental impacts of buildings, consciousness should be gained about all the factors involved, so it becomes necessary to make an LCA evaluation, already in the design phase. This LCA

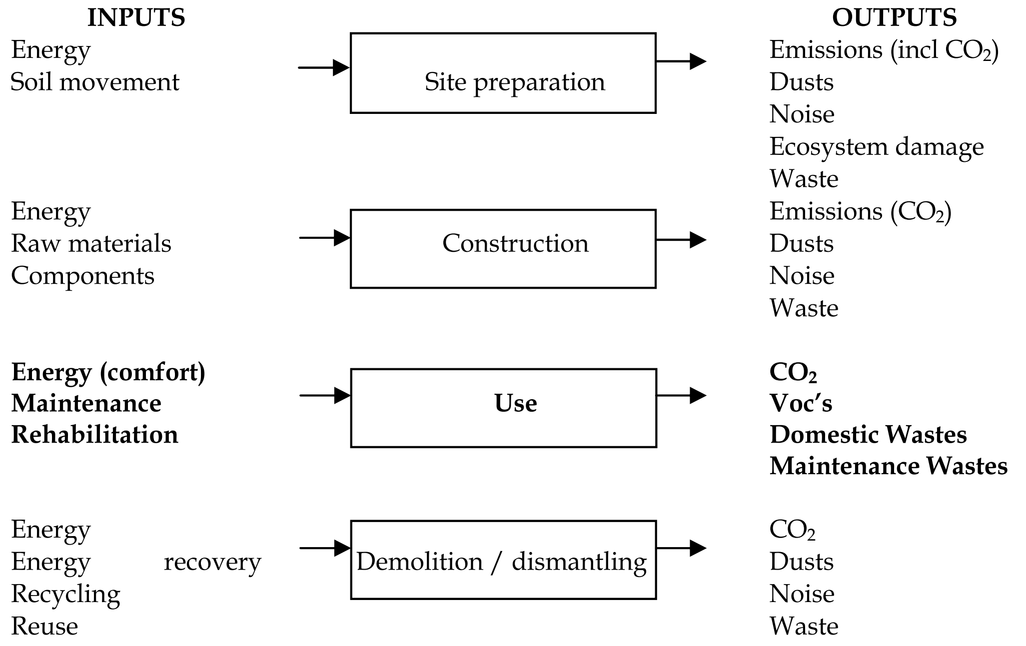

Fig. 4. Environmental Impact of buildings in its Life Cycle

evaluation should consider closed-loop systems, as represented in Figure 5. In the scheme of Figure 4 are marked in bold the inputs and outputs corresponding just to the use phase, in a close loop cycle. When building is designed for deconstruction, reuse or refurbishing beyond it's expected lifecycle, only these impacts remain present.

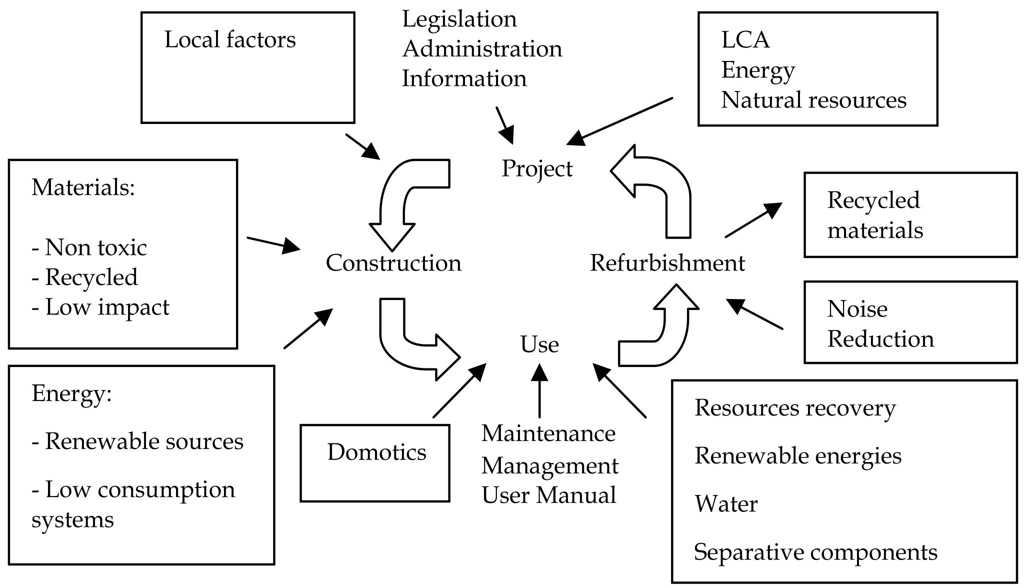

Fig. 5. Life cycle of buildings in Closed Loop – adapted from Mendonça (2005)

The impacts that building construction has on the environment can be analysed from the following points:

- Position and integration of buildings in the site;
- Influence of design in the Building behavior during its useful life;
- Influence of the equipments in the Building behavior during its useful life;
- Characteristics of the materials used – by the impact that these can produce on the environment during the processes of extraction of raw materials, manufacture, useful life and in the end of life scenarios (reuse / recycling / energy recovery).

2.1 Energy fluxes of buildings

The energy component of the building construction is not only related with the stages of extraction and production of materials and work, but continues through the use of the building and even during the demolition, so the overall environmental impact assessment of a building becomes complex. It is therefore relatively difficult to differentiate the energy component from the material component, as in virtually all phases of the building life cycle the two components are present.

According to Dimson (1996), buildings account for 40% of the energy consumed annually. These values were calculated for buildings located in central and northern Europe. In Portugal, the mild climate and a situation of generalized discomfort inside buildings has meant that the consumption associated with the heat and cooling needs - about 20% of total energy consumption - has not, in relative terms, nothing to do with the levels of consumption in northern Europe countries (Mendonça, 2005).

In relation to the overall percentage of energy consumption during 50 years of use, the amount of energy that actually goes into the production of construction materials in a building, is between 6 and 20% and depends on building type, climate, etc. (Berge, 2000). The intervention in reducing the embodied energy of the materials is much more significant in overall energy consumption than in countries with less favorable climate, so it can be concluded that this factor has greater importance in Portugal than in most other European countries.

Energetic consumption in the demolition and removal of building wastes constitutes in average around 10% of the total energy spent since its production (Berge, 2000), so the attitude of those who conceive the buildings should consider that energetic cost can still be amortized after the 50 years generally considered for the useful life, reusing or at least recycling as much as possible in the end of this period.

Energy use in buildings is divided between production, distribution and use of building materials, as summarized in Figure 6.

The manufacture, maintenance and renewal of materials in a housing building made of concrete blocks, for a lifetime of 50 years, require an energy consumption of 3000MJ/m^2. For larger buildings, in steel or reinforced concrete, the energy required is approximately 2500MJ/m^2 (Berge, 2000).

The embodied energy of a material corresponds to the energy used to manufacture a product. It corresponds in average to 80% of the total amount of energy associated to final product installed in the building. Embodied energy is divided as following (Berge, 2000):

- Direct energy consumption due to the extraction of raw materials and manufacturing process. It varies with the manufacturing system and the type of equipments used;
- Indirect energy consumption from the manufacturing process. It refers to the energy consumption of equipment, air conditioning and lighting in the factory, and is usually a value less significant than the direct;

- Transport energetic costs, of raw materials and semi-processed materials. The choice of transport system used is also a decisive factor. The road transport is one of the most inefficient, it implies over 400kWh/kg.Km, and this is the most used transport in the Portuguese case.

Energy fluxes in buildings

- Materials
  - - Embodied energy
    - - Direct consumption (extraction of raw materials and manufacture)
    - - Indirect consumption (consumption of the production unit)
    - - Raw materials transport
  - - Transport of products
- Construction
  - - Direct consumption
    - - Consumption with equipments
    - - Consumption with hand labor
  - - Indirect consumption
    - - Transport of personnel
    - - Transport of equipments
    - - Manufacture and maintenance of equipements
- Use
  - - Maintenance
    - - Cleaning
    - - Refurbishment
  - - Confort
    - - Lighting
    - - Climatization
    - - Ventilation
- Demolition
  - - Dismantling
  - - Transport of materials to landfill or recycling

Fig. 6. Energetic fluxes in buildings – adapted from Mendonça (2005)

Massive CO_2 emissions caused by combustion engines are related with the construction industry, in large part associated to the transportation of construction materials, as well as labors. In the case of construction materials, the random location of works, the preferred mean of transport is road.

The energy pollution in the manufacturing process of a given material depends on the type and quantity of primary energy spent. Energy sources vary from country to country but in Portugal, the most commonly used types of energy are fossil fuels. The construction materials of higher embodied energy may thus contribute indirectly to the increased CO_2 and other pollutants emissions.

2.2 Material fluxes of buildings

The material environmental impact of buildings is essential due to raw materials extraction. The construction industry is the second largest consumer of raw materials in the world today, after the food industry (Berge, 2000). The building industry is responsible for consuming 25% of wood production and 40% of aggregates (stone, gravel and sand) around the world. Buildings are also responsible for 16% of water consumed annually (Dimson, 1996).

Material pollution is related mainly to pollutants in air, land and water from the material itself and from the others components of the material when in production, use and demolition. The picture becomes more complex considering that about 80,000 chemicals harmful to health, are used in the construction industry, and that their number has quadrupled since 1971 (Berge, 2000). In Table 2 are shown the types and quantity of waste associated with building materials production.

Most material environmental impacts are due to the exploration of the non-renewable raw materials resources, particularly minerals and aggregates. Quarries and opencast mines, as well as the extraction of sand, produce visual impacts on the landscape, destroy ecosystems and pollute the soil waters. The pollutants concentration percentage in the wastes resulting from demolition of buildings is relatively small; however, as the amount of waste produced is very high, this represents a substantial part of the overall environmental impacts. A great percentage of the building construction wastes in Portugal (concrete and brick) are not in general treated or selected for reuse or recycling, being only used as inert for land filling in sanitary or industrial municipal landfills.

The losses in construction are approximately 10% of the total losses in the construction industry (Berge, 2000). Each material has a loss coefficient that describes the waste during storage, transportation and installation of the final product. For many materials, increased pre-fabrication does decrease this factor, as well as the standardization of products and building design taking these factors into account.

In the construction industry, a large amount of packaging is used in the transportation and storage of products. An important aspect of packaging should be its easy recycling or even reuse.

3. Waste management in building construction

In Portugal and southern Europe in general, the heavyweight building systems made of concrete structure and hollow brick, increasingly hinders reuse, in opposition to what should be expected. Interestingly, the buildings with more than 50 years, present more easily reusable components, and have an initial much lower environmental impact. In these buildings, systems were simple, often with juxtaposed stone masonry walls, timber pavement and roof structures with ceramic tiles. Even in northern Europe, more sensitive to environmental aspects, this phenomenon is a reality. Selective demolition of buildings, where a level of recycling of 90% was achieved, is only possible in old buildings, using fewer materials and well differentiated (Berge, 2000). According to Berge, it is doubtful that the level of recycling can reach even 70% in newly constructed buildings, even in northern Europe realities. This is mainly due to the extensive use of composite elements, with aggregate materials. For example, in steel reinforced concrete, where steel content can reach 20%, recycling of the metal is a relatively complex process, due to the need of separating the two elements, which can result economically unfeasible in most cases.

3.1 Implementing a waste minimisation hierarchy

Waste management can be hierachically classified in three levels, by decreasing order of effectiveness:

- Reuse;
- Recycling;
- Energy recovery.

Material		Wastes from materials production process: g/kg of product	Wastes from materials production process: Taken to special landfills (%)	Wastes from building construction/ demolition: Waste types*
Steel	100% recycled			D
	galvanized (from mineries)	601	5	D
	stainless (from mineries)			D
Chipboard	porous without bitumen	81	5	A/D
	porous with bitumen			B/E
	high density without bitumen	80		A/D
	high density with bitumen			B/E
Aluminium (50%recycled)		715	20	D
Concrete (with Portland cement)	structural	32		C
	fibre reinforced slabs	81	10	C
	mortar	17	10	C
	lightweight aggregate blocks	58	13	C
Bitumen		3		B/D
Lead (from ore)		265	5	E
Polyvinyl Chloride (PVC)				D
Copper (from ore)		2.410	84	D
Maritime counterplate		40	2	B/D
Cork				A/D
Cellulose fibre	100% recycled w/ boric salts			E
	paper 98% recycled			A/D
Carton plaster		8	10	D
Rockwool		320	5	D
Glasswool		90	5	D
Linoleum		2		B/D
Timber	non treated	25		A/D
	treated			E
	glulam			B/D
Ceramic tiles		9		C
Stone				C
Polyester (UP)				B/D
Expanded Polystyrene (EPS)				B/D
Extruded Polystyrene (XPS)				B/D
Expanded polyuretane (PUR)		486	7	B/D
Expanded perlite	with bitumen			E
	without bitumen			C
Compacted earth				C
Clay brick		87	15	C
Glass				C

* A – Burn without filtering; B – Burn with filtering; C – Landfill or inert; D – Municipal landfill; E – Special landfill.

Table 2. Wastes associated to manufacture and building industries. Source: (Berge, 2000)

The management should preferably be developed in order that materials can be returned in its original quality level and not at an inferior level - “downcycled” (Berge, 2000).
The reuse of materials after the demolition should be taken into account. The reuse depends on component useful life and refers to the use responding to the same function. An effective reuse of building components requires simplified and standardized products, which almost never happens. However, reuse of materials has been a fairly common construction practice. In coastal areas, some buildings were constructed using materials recovered from dismantled ships. The prefabricated building in timber is therefore an example of construction with a high potential for reuse. In some coastal areas of Portugal, vernacular buildings are made in this system.
Recycling, rather than manufacturing products from natural raw materials can substantially reduce their environmental impacts. A product that can easily be reused several times has advantages over lower cost products that can not be reused. In Portuguese building industry, products present high durability but low potential for recycling, but what is more problematic, there are products with low durability and great recycling potential that are not usually recycled.
Applying to few contemporary building components, but to many old building components, energy recovery is also possible as a last option. But this can only be beneficial if this energy is extracted in a site near the building, but also if the combustion process can be kept clean.
The waste minimisation hierarchy is an important guide to managing waste. It encourages the adoption of options for managing waste in the following order of priority (Morgan & Stevenson, 2005):

- Waste should be prevented or reduced at source as far as possible;
- Where waste cannot be prevented, waste materials or products should be reused directly, or refurbished before reuse;
- Waste materials should then be recycled or reprocessed into a form that allows them to be reclaimed as a secondary raw material;
- Where useful secondary materials cannot be reclaimed, the energy content of waste should be recovered and used as a substitute for non-renewable energy resources; and
- Only if waste cannot be prevented, reclaimed or recovered, it should be disposed of into the environment by landfilling, and this should only be undertaken in a controlled manner.

In Figure 7 is illustrated the waste hierarchies for demolition and construction operations.
Construction waste management should move increasingly towards the first of these options, using a framework governed by five key principles promoted by the European Union (Hurley and Hobbs, 2004):

- The proximity principle;
- Regional self sufficiency;
- The precautionary principle;
- The polluter pays; and
- Best practicable environmental option.

Clearly, the reuse of building elements should take priority over their recycling, wherever practicable, to help satisfy the first priority of waste prevention at source.
To ignore deconstruction means to create a pile of debris that cannot be viably reused. The Figure 8 attempts to depict this situation; to demolish a building without resorting to procedures that enable separation and recovery of debris and by-products.

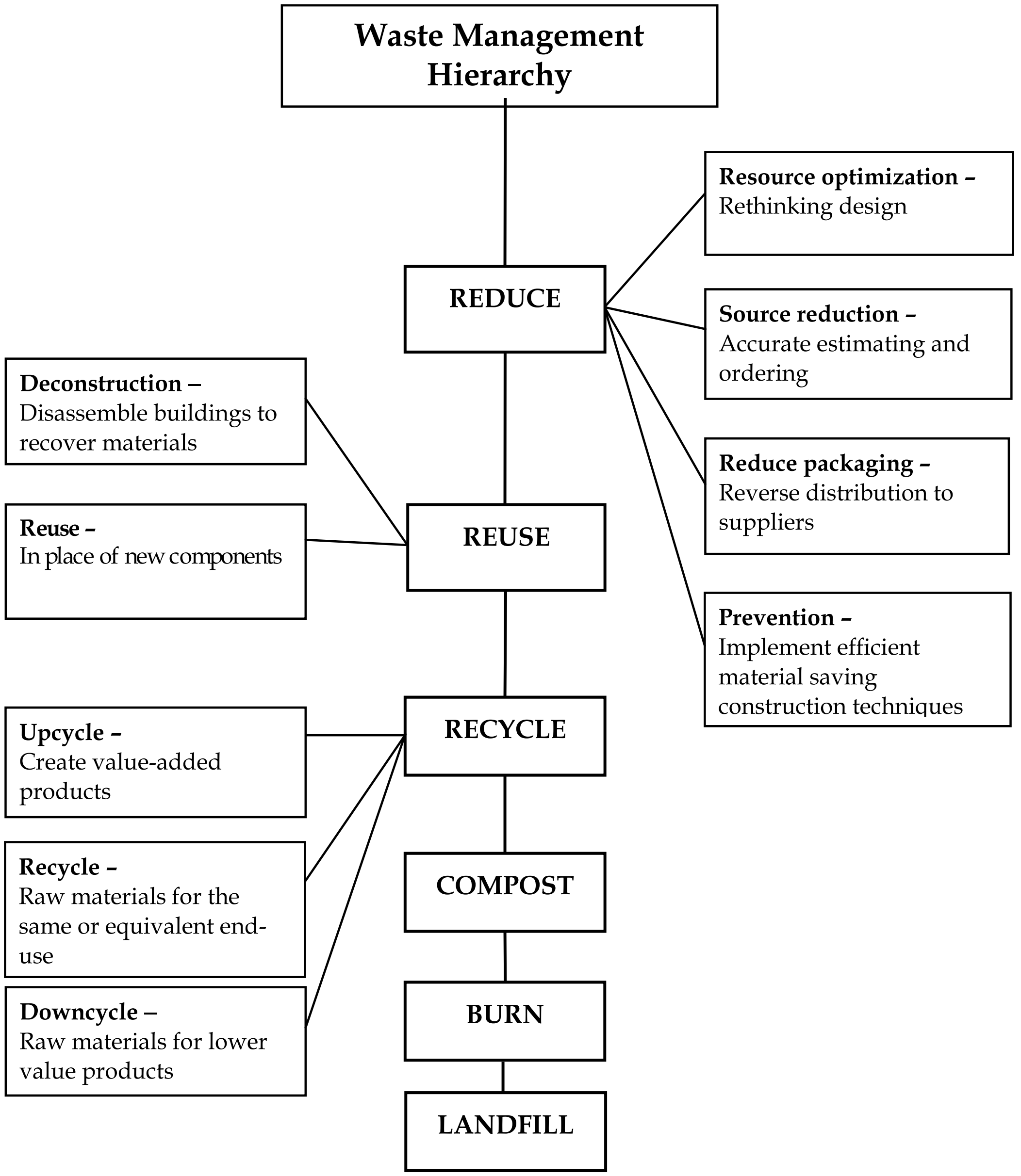

Fig. 7. Hierarchies for demolition and construction operations. Source: Adopted directly from (kibert & Chini, 2000)

Fig. 8. Sample of an undifferentiated demolition. Source: (Pinto, 2000)

The Figure 9 attempts to depict that deconstruction permits the resorting to procedures that enable separation and recovery of debris and by-products.

Fig. 9. Sorted broken concrete and steel stockpiled separately (Public Fill Committee, 2004)

The benefits from reuse are significant. The main benefits of building reuse include sustainability, direct and indirect monetary savings, an accelerated construction schedule, and decreased liability exposure (Fig. 10).
Although the reuse can benefit all projects, the situation more clearly advantageous for the reuse of construction is in urban environments, because the construction sites can be close to existing buildings and cause negative impacts on surrounding ((Chapman et al., 2003) cited by (Laefer & Manke, 2008)).
Building deconstruction supports the waste management hierarchy in its sequence of preferred options for the management of generated C&D waste materials (see Figure 7). If a building is still structurally sound, durable and flexible enough to be adapted for a different use (either in situ or by relocation), then waste can be *reduced* by *reusing* the whole building. If components and materials of a building can be *recovered* in high quality condition,

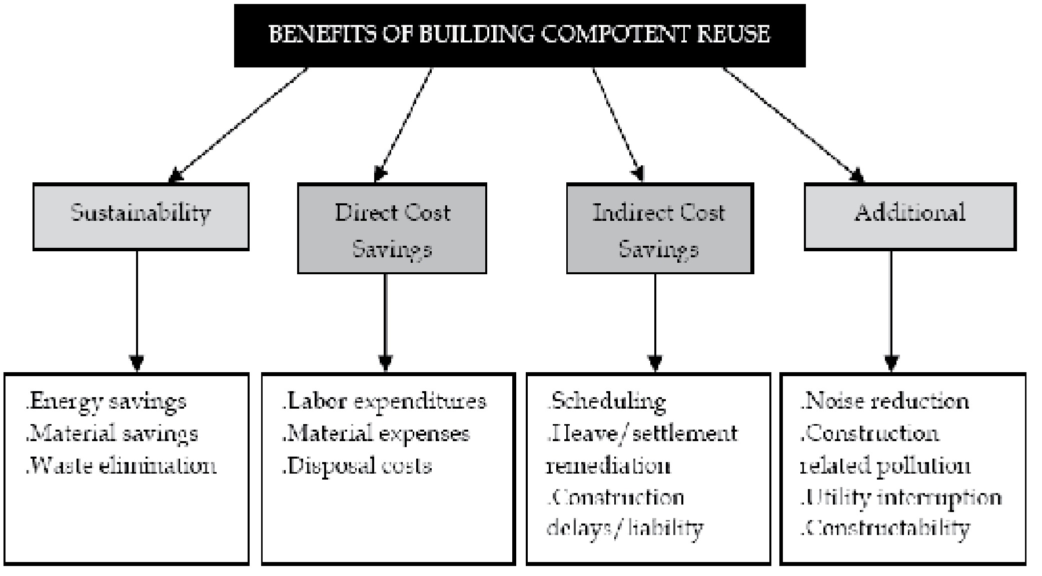

Fig. 10. Benefits of building component reuse. Source: Adopted directly from (Laefer & Manke, 2008)

then they can be *reused*. If the building materials are not immediately reusable, they can be used as secondary feedstock in the manufacture of other products, *i.e.*, *recycled*. The aim is to ensure that the amount of waste that is destined for *landfill* is reduced to an absolute minimum. This approach closes the loop in material flow thereby contributing to resource efficiency.

4. Deconstruction as alternative to traditional demolition process

4.1 Barriers and advantages of deconstruction

There are a number of areas where the authorities may influence design and planning strategies at an early stage. These include fiscal incentives such as the maintenance of a fixed price for recovered products or increased costs for waste disposal through the landfill tax. Incorporation of deconstruction techniques into material specifications and design codes on both a National and European level would focus the minds of designers and manufacturers. Education on the long-term benefits of deconstruction techniques for regulators and major clients, would provide the necessary incentive for the initial feasibility stage. Design for deconstruction is not, however, solely an issue for the designers of buildings. The development of suitable tools for the safe and economic removal of structural elements is an essential pre-requisite for a more widespread adoption in deconstruction (Couto & Couto, 2007).

A study carried out by BRE (Building Research Establishment) (Hurley et al., 2001) has shown what the industry has known for decades; that there are keys factors that affect the choice of the demolition method and particular barriers to reuse and recycling of components and materials of the structures. The most factors are physical in terms of the nature and design of the building along with external factors such as time and safety. Future factors to consider should well include the fate of the components, the culture of the demolition contractor and the 'true cost' of the process. For the latter, barriers to uptake include the perception of planners and developers, time and money, availability of quality information about the structure, prohibitively expensive health and safety measures, infrastructure, markets quality of components, codes and standards, location, client perception and risk.

According to Hurley and Hobbs (2004), the main barriers (in the UK) to the increased use of deconstruction methods within construction include:

- Lack of information, skills and tools on how to deconstruct;
- Lack of information, skills and tools on how to design for deconstruction;
- Lack of a large enough established market for deconstructed products;
- Lack of design. Products are not designed with deconstruction in mind;
- Reluctance of manufactures, which always prefer to purchase a new product rather than to reuse an existing one;
- Composite products. Many modern products are composites which can lead to contamination if not properly deconstructed or handled;
- Joints between components are often designed to be hidden (and therefore inaccessible) and permanent.

Although the market for products from deconstruction is poorly developed in Portugal, can be noted that the interest in low volume, high value, rare, unique or antique architectural components is much higher than the interest in materials that have high volume, low value, such as concrete.

Even though there are significant advantages to deconstruction as an option for building removal, there are still more challenges faced by this alternative:

- Deconstruction requires additional time. Time constraints and financial pressure to clear the site quickly, due to lost time resulting from delays in getting a demolition, or removal permit, may detract from the viability of deconstruction as a business alternative;
- Deconstruction is a labor-intensive effort, using standard hand tools in the majority of cases. Specialized tools designed for deconstructing buildings often do not exist;
- The proper removal of asbestos-containing materials and lead-based paints, often encountered in older buildings that are candidates for deconstruction, requires special training, handling, and equipment;
- Re-certification of used materials is not always possible, and building codes often do not address the reuse of building components.

The main opportunities which require development include:

- The design of joints to facilitate deconstruction;
- The development of methodologies to assess, test and certify deconstructed elements for strength and durability, etc.;
- The development of techniques for reusing such elements;
- The identification of demonstration projects to illustrate the potential of the different methods.

Modern materials such plywood and composite boards are difficult to remove from structures. Moreover, new building techniques such as gluing floorboards and usage of high-tech fasteners inhibit deconstruction. Thus, buildings constructed before 1950 should be ideally targeted for deconstruction (Moussiopoulos et al., 2007). In Portugal, it is expected a substantial increase in the investment on refurbishment of buildings. The deconstruction should have a relevant contribution in this process.

The greatest benefit will be achieved by incorporating deconstruction issues into the design and feasibility stage for all new construction. Each case can then be judged on its merits in terms of the potential cost of recovery and recycling or reclamation and reuse of construction materials.

4.2 Deconstruction benefits

Deconstruction seeks to close the resource loop, in order that existing materials are kept in use for as long as possible and the deployment of new resources in construction projects is diminished. The benefits from deconstruction are considerable. Deconstruction offers historical, social, economic and environmental benefits. Older buildings often contain craftsmanship which have significant historical value. Deconstruction can carefully salvage these important historical architectural features, because materials are preserved during removal. Deconstruction is more time consuming and requires more skill than simply demolishing a structure. Although the extra time required could act as a detriment, deconstruction provides training for the construction industry and also has the potential to create more jobs in both the demolition and the associated recovered materials industry. Deconstruction provides a market for labour and sales of salvaged material. More important, deconstruction puts back into circulation items which may be directly used in other building applications. Environmental benefits of deconstruction are essentially two fold. Primary, resource use is reduced through a decreased demand on new materials for building. This means that climate change gas emissions, environmental impact, pollution (air, land and water) and energy use are all reduced. Deconstruction also means that less waste goes to landfill because materials are salvaged for reuse. This means fewer new landfills or incinerators need to be built which reduces the environmental and social impact of such facilities, and environmental impact of existing landfills is reduced. Currently there are few incentives to break the historical practice of landfilling debris. The occasionally higher cost of selected demolition can be offset by the increased income from salvaged materials, decreased disposal costs, and decreased costs from avoided time and expense needed to bring heavy equipment to a job site (Couto & Couto, 2007).

Based on the review of international literature it is possible to categorize the main benefits of deconstruction as follows:

- Reuse and recycle materials: materials salvaged in a deconstruction project can be reused, remanufactured or recycled (turning damaged wood into mulch or cement into aggregate for new foundations) (Hagen, 2008);
- Foster the growth of a new market — used materials: recovered materials can be sold to a salving company. The market value for salvaged materials from deconstruction is greater than from demolition due to the care that is taken in removing the materials in the deconstruction process;
- Environmental benefits: salvaging materials through deconstruction helps reducing the burden on landfills, which have already reached their capacity in many localities. By focusing on the reuse and recycling of existing materials, deconstruction preserves the invested embodied energy in materials, eliminating the need to expend additional energy to process new materials. By reducing the use of new materials, deconstruction also helps reducing the environmental effects, such as air, water and ground pollution resulting from the processes of extracting the raw materials used in those new construction materials. Deconstruction results in much less damage to the local site, including soil and vegetation, and generates less dust and noise than demolition;

- Create jobs: deconstruction is a labour-intensive process, involving a significant amount of work, removing materials that can be salvaged, taking apart buildings, and preparing, sorting, and hauling the salvaged materials.

Other less obvious benefits may also come from the deconstruction, but that depend on the specific characteristics of countries and regions.

4.3 Cost of deconstruction

Deconstruction, as an environmentally-sound business practice, is not necessarily more costly than traditional demolition. Buildings can be often deconstructed more cost-efficiently than they can be demolished. There are many different factors involved, including the type of construction and the value of the materials that can be recovered. But overall, deconstruction can be more cost-effective than demolition. Not only can buildings be deconstructed more cheaply than they can be demolished, but deconstruction provides construction companies with low-cost materials for reuse in their own building projects. Deconstruction is also an ideal training ground for the construction trades. Preliminary results from pilot projects carried out in different parts of the USA by the US Environmental Protection Agency (EPA) have indicated that deconstruction may cost 30 to 50% less than demolition (CEPA, 2001).

Deconstruction is labor-intensive, involving a higher level of manual work than there would be in a demolition project. But the higher labor cost can be offset by lower costs for equipment rent and energy usage, cost savings in the form of lower transportation and landfill tipping charges, and the revenues from sales of the salvaged material.

Research shows that the market value for salvaged material is greater when deconstruction occurs instead of demolition, because of the care taken in removing materials. Money made through salvaging can be used to offset other redevelopment costs. Lastly, disposal costs are lower with deconstruction because the process reduces the amount of waste produced by up to 75 percent.

Different studies carried out in Germany on deconstruction methods have showed that optimized deconstruction combining manual and machine dismantling can reduce the required time by a factor of 2 with a recovery rate of 97% (Kibert, 2000). In the Oslo region, Norway, it is estimated that between 25% and 50% of C&D waste stream is recycled or reused (Kibert, 2000).

In Portugal the construction waste management is now beginning its first steps, so, its outcomes are not yet completely known.

Previous research analysis point out that from the clients' perspective the following are sound economic reasons for using deconstruction (Couto & Couto, 2009):

- To increase the flexible use and adaptation of property at minimal future cost;
- To reduce the whole-life environmental impact of a project;
- To maximise the value of a building, or its elements, when it is only required for a short time;
- To reduce the quantity of materials going to landfill;
- To reduce a future liability to pay higher landfill taxes;
- To reduce the risk of financial penalties in the future, due to changing legislation, through easily replaceable building elements;
- To minimise maintenance and upgrading costs incurred by replacement requirements.

A key economic benefit of design for deconstruction is the ability for a client to "future proof" their building, both in terms of maintenance and any necessary upgrading, with minimum disruption and cost. The wider economic benefits to society include minimising waste costs at all levels.

Numerous projects have been costed, and while some have come in on budget, others have not. Much depends on the canniness of the design team and contractor, from the outset, with cost savings to be viewed as bonus rather than a given. Design for deconstruction should always be adopted for its wider economic, social and environmental benefits rather than any initial cost saving.

Current economic barriers to design for deconstruction and reuse of reclaimed materials and products include: the additional time involved for deconstruction and the difficulty of costing this against reused materials which will be used on a different project, the damage caused by poorly designed assemblies and connectors, as well as the limited flexibility of reclaimed elements. Reuse is not subsidised in the same way that manufacture is in terms of energy, infrastructure, transportation, and economies of scale, all of which have hidden environmental costs.

5. Designing for deconstruction

In the concept of construction management, building towards a future scenario of deconstruction is an important factor. With this concept, the different components can be easily separated during the demolition, separating the components of each type for reuse, but also facilitating recycling and energy recovery (Berge, 2000).

Addis & Schouten (2004) synthesized the following deconstruction design strategies to facilitate reuse and recycle:

- Use materials that can easily be recycled;
- Use materials for which, when recycled, a viable market exists;
- Whenever possible design products or elements that can be separated easily into units made of one material;
- Whenever possible design products or elements whose materials all decay at the same rate, so they reach their end of the life simultaneously;
- Ensure that materials, once deconstructed and separated, are clean and free from contamination and paint – this will maximize their reusability or recyclability, although it may compromise their durability;
- Use alternatives to chemical bonding (adhesives) in favour of bolts, clips, etc.

A summary of strategies that can adapt to the Portuguese and thus allow to complete a draft prepared for the deconstruction consists in:

- Using totally separated systems;
- Possibility to separate components in each system;
- Using standardized and homogeneous materials.

5.1 Separated building constructive systems

A building is composed of various building components, forming systems (structure, facades, fittings, partitions, furniture, etc.). The structural system has to last the entire lifetime of the building, while interior partitions are often rearranged in short periods of time, for functional or more futile reasons.

In Portuguese contemporary buildings of conventional construction, the different systems are almost always permanently fixed, forming an inseparable unit, which causes that components with short useful life may condition components with long useful life, which is unwise when the smaller durability component is for example the structure. It becomes common, for example, to demolish buildings where facilities are integrated in the structure and thus it became difficult to maintain or replace. A fundamental principle for efficient reuse of building components is the differentiation of the systems. Figure 11 presents examples of three types of connection between wall and structure: the image (a) show the connection between walls and structure, which was the common situation in the buildings in Portugal until about 50 years; the image (b) show the common situation today with brick masonry walls and reinforced concrete structure; and image (c) show the situation in separate systems, whose materials can be of the same quality or not, but always easily separable.

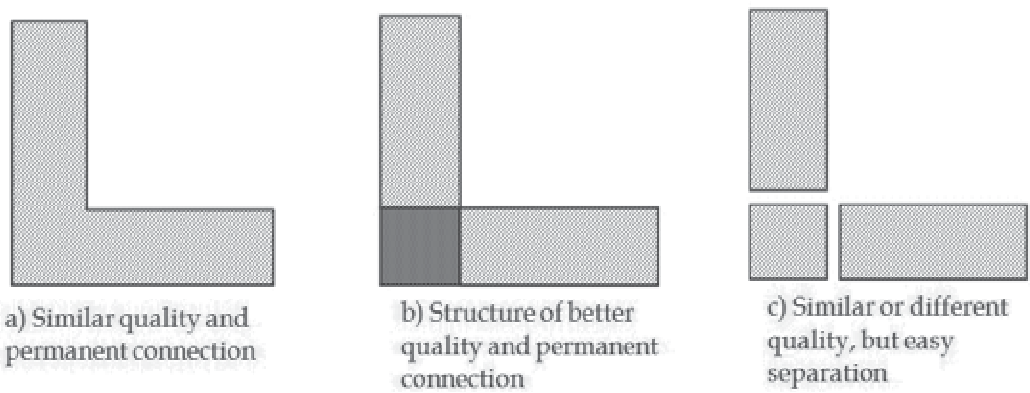

Fig. 11. Connections between structural and wall systems. Adapted from Berge (2000)

Easily dismantling building systems should comprise components prepared to be loose fitted together during assembly and are commonly known as prefabricated. The prefabricated lightweight systems present as a main advantage to be easily transported in cargo volume and small weight, potentially making them easier to move over large distances. In places with difficult access to large transport vehicles, these represent a constructive solution economically more feasible than the conventional heavyweight one. It starts to be common in Portugal, mainly for single family houses, and marketed by companies that normally are responsible for their design and assembly. The most common material used is timber, although metal frames and sheets are also common options.

5.2 Durability and possibility to separate the systems' components

From the standpoint of material resources, there is always a clear advantage in using more durable materials for buildings, allowing the longest lifetime possible (Berge, 2000). The use of durable materials allows reducing the raw materials used, since ensuring durability equal to all components of the same building system, so as not to compromise the durability of materials by the existence of lower durability. If it is impractical to use materials of equal durability, the type of material, then the replacement of less durable materials should be easier. The building layers model of Brand (1995) allows to understand and manage the different components in relation to its durability.

Durability depends from diverse factors, such as:

- The material in itself, by its physical and chymical structure;
- Building and execution, where and how the material is placed;
- Local environment exposure - sunlight, raining, pollutants and other conditions.

Components of each system should be easily divided into units for easy handling, allowing reuse and recycling. Separation allows easy substitution of elements with greater wear; easy replacement of elements after repair; and reuse elements in areas of less visual exposure in exchange for the elements with less wear. It also allows the easy transport of components within the building itself and outside it.

5.3 Standardized and homogeneous materials

Many building components are composed of different materials combined in a new material with different and increased properties, often called composite. But the reuse or recycling of composite materials is often impossible or very difficult. On the other hand, different degrees of durability of the materials present within the same component can result in a material that can reach the loss of its useful life, while others are still valid, but it is no longer possible to use the component for that reason (Berge, 2000).

The use of homogeneous materials, such as hardwood timber in a floor or natural stone in a wall, allows re-use later, fulfilling the same purpose, something not possible with the use of most composites. For example, between an outer coating in corrugated iron or a plastic composite sandwich panel, the last one is unlikely to be reused and recycled while in the first case any of these hypothesis is feasible.

6. Conclusion

All around the world, the deconstruction of buildings has gained more and more attraction in recent years as an important waste management tool. Deconstructing a building consists on the careful dismantling of their components, so as to make possible the recovery of materials, promoting reuse and recycling. The concept arose as a consequence of the rapid increase in the number of demolished buildings and the evolution of environmental concerns within society at large. In fact, demolition is one of the main construction activities in what concerns to the production of waste. The deconstruction is an unusual process in Portugal; as traditional demolition is yet the preferred method when it is necessary to dismantling a building. In addition to the general lack of awareness about the overall benefits of deconstruction, there are many barriers to deconstruction in Portugal. The barriers have many sources that include not only technical and market issues, but also issues related with social and educational factors. The barriers to the implementation of deconstruction were disclosed as well as its opportunities.

Strategies and actions that could be implemented in Portugal by impelling the deconstruction process were discussed in order to improve waste construction management. The focus was on easy to implement design for deconstruction strategies, having in view the prediction of future scenarios of deconstruction. To achieve this goal, the different components should be easily separated during demolition, allowing its reuse, and if this is not possible, at least allowing the recycling or even the energy recovery.

Various factors allow achieving a deconstruction effective project, such as: using totally separated systems; Possibility to separate the components in each system; Using standardized and homogeneous materials; Using mechanical or dry joints; Use lightweight materials and components. These strategies can make handling easier, quicker, and less costly, thereby making reuse a more attractive option.

In Portugal, recent legislation about waste management in construction has come into force, but is still giving its first steps and there are still many difficulties to overcome. There are some good examples but these are still insufficient.

Therefore, a greater engagement and a new attitude from all practitioners is absolutely necessary in order to implement new and more adequate waste management rules and new selection demolition processes so as to increase the results of the construction waste management.

It is very important that National authorities and construction practitioners understand the benefits of the deconstruction process and look at it as an advantageous way to improve waste management, thus following other European countries' practices.

7. Acknowledgment

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17

Hydraulic Conductivity of Steel Pipe Sheet Pile Cutoff Walls at Coastal Waste Landfill Sites

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1. Introduction

Landfill sites are facilities where the final residue is disposed after all possible recycling energy has been recovered from it. Therefore, landfill sites are an important part of civil infrastructure, required for environmental conservation without dumping waste in residential areas. However, in many cases, the construction of landfill sites has been opposed due to concerns of residents living the vicinity regarding environment safety with regard to situations such as "the leachate from waste may leak out" ; hence, the construction of new landfill sites has become more difficult. Moreover, the construction cost of landfill sites has also significantly increased simultaneously due to tighter environmental legislation (Shimizu, 2003; Kamon et al., 2007).

In Japan, small-scale inland landfill sites were often constructed in the river-head areas of mountain valleys. With regard to the abovementioned social concerns regarding the landfill sites, the locations of landfills have recently been diversified into coastal areas on a large scale. These sites are developed at urban harbour areas in order to reduce the risk of contaminating the groundwater, which can be caused by the leakage of leachate, and conserve the water resources (Kamon & Inui, 2002). In the national statistics of 2003 announced at Ministry of the Environment, the capacity of coastal landfill sites was 23.3% of that of all landfill sites, and particularly in metropolitan areas, it was greater than 80% (see Fig. 1). These statistics indicate that the role of coastal landfill sites has been increasing steadily. However, the residents living in the vicinity of these sites continue to express the same concerns for environment safety. Therefore, ensuring stable and systematic operation of the coastal landfill sites in the future and prolonging the life of coastal landfill sites constructed until now are important matters of concern, particularly in metropolitan areas.

A revetment at a coastal landfill site ensures space for waste disposal and harbour maintenance during the disposal of waste, construction sludge, dredged soil etc. A revetment at a coastal landfill site must function as a vertical (side) cutoff barrier that prevents the leakage of leachate containing toxic substances from the landfill waste, into the sea; furthermore revetments must protect the coastal landfill site from various external forces such as earthquakes, ocean waves, high tides and tsunamis (Waterfront Vitalization and Environment Research Center, 2002).

Recently, steel pipe sheet piles (SPSPs), using which the deepwater construction is possible (Japanese Association for Steel Pipe Piles, 1999), have been widely employed in vertical cutoff barriers at coastal landfill sites due to their workability and economical efficiency. A

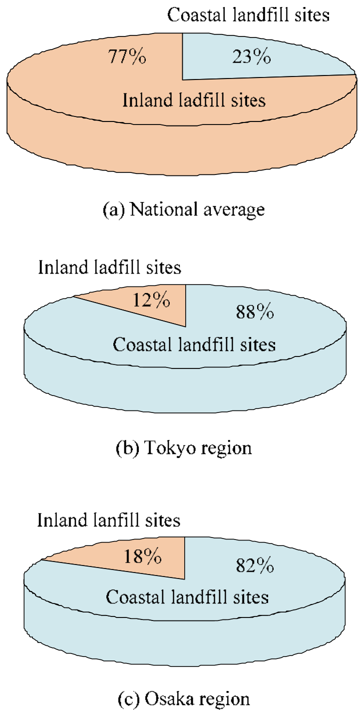

Fig. 1. Capacity comparison between inland and coastal landfill sites based on national statistics of 2003 announced at Ministry of the Environment, Japan itle of figure, left justified

vertical cutoff barrier employing SPSPs is called a "SPSP cutoff wall" in this study. However, the design and application of SPSP cutoff walls, evaluation of environmental feasibility, construction technology and long-term maintenance are very complicated both experimentally and analytically (Kamon et al., 2001). This is because of the existence of joint sections in the SPSPs, as shown in Fig. 2.

The appropriately estimation of the hydraulic performance of SPSPs with joint sections (shown in Fig. 2) is an important issue, particularly in the evaluation of environmental feasibility, that is, the containment of leachates containing toxic substances. Figure 3 shows the characterization of the environmental feasibility of vertical and bottom cutoff barriers as well as the overall landfill site. When evaluating the hydraulic performance of an SPSP cutoff wall, an equivalent hydraulic conductivity is generally obtained (Waterfront Vitalization and Environment Research Center, 2002). This equivalent hydraulic conductivity assumes that the joint section and the steel pipe are integrated; therefore, the hydraulic conductivity is substituted with a uniform permeable layer (see Fig. 4). The Prime Minister's Office and the Ministry of Health and Welfare says that the integrated equivalent hydraulic conductivity with 50 cm thickness must be 1.0×10^{-6} cm/s or less (Waterfront Vitalization and Environment Research Center, 2002). However, in an evaluation that employs the equivalent hydraulic conductivity, it is difficult to consider the local leakage of leachate containing toxic substances from the joint sections in the SPSP cutoff wall.

In this study, an evaluation method that can express the local leakage of leachate from the joint sections in the SPSP cutoff walls is discussed. In particular, the evaluation of the environmental feasibility (containment of leachates containing toxic substances) considering a

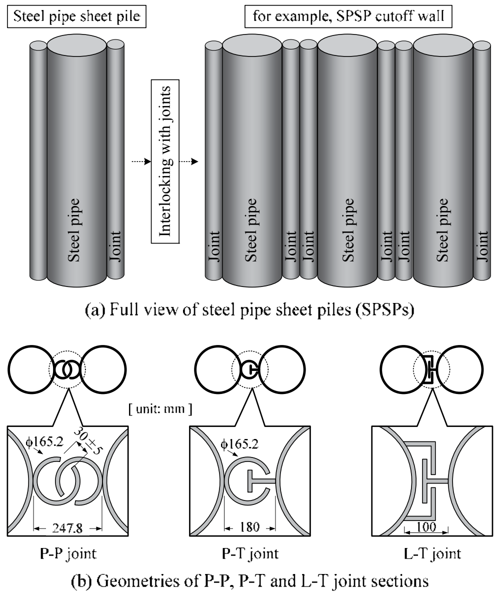

Fig. 2. Schematic diagram of steel pipe sheet piles with joint sections

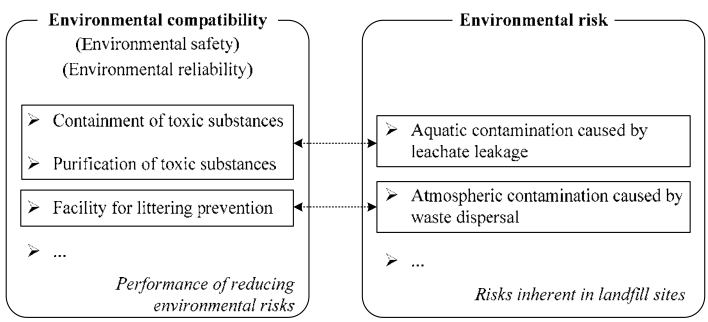

Fig. 3. Characterization of environmental feasibility on vertical and bottom cutoff barriers as well as overall landfill site

three-dimensional arrangement and hydraulic conductivity distribution of the joint sections in the SPSP cutoff wall is compared with an evaluation that uses the equivalent hydraulic conductivity.

2. Analysis for environmental feasibility

The development of methods for the detection of the generation points of leachate leakage has been conducted in various different ways at inland and coastal landfill sites in order to determine when the leachate containing toxic substances will leak into the surrounding areas after the land has been reclaimed at the landfill site (Kamon & Jang, 2001; The Landfill System & Technologies Research Association of Japan, 2004). However, the present detection methods are insufficient with regard to their durability, and the use of these methods may lead to excess cost and time for repairing the generation points of leachate

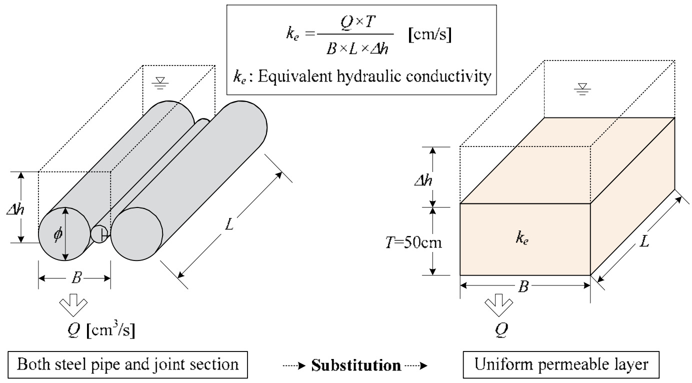

Fig. 4. Concept of equivalent hydraulic conductivity assuming that joint section and steel pipe are integrated

leakage in the vertical and bottom cutoff barriers at the landfill sites. Therefore, an effective implementation and verification of the seepage and advection/dispersion analysis, considered as a two-dimensional or a three-dimensional problem, of the leaching behavior of leachate containing toxic substances are necessary along with the upgradation of the technique used to repair vertical and bottom cutoff barriers. The structure of vertical and bottom cutoff barriers that can ensure long-term stability as well as the evaluation method for the environmental feasibility of landfill sites must be also discussed.

The leaching behavior of leachates containing toxic substances near the vertical and bottom cutoff barriers at landfill sites must be considered with regard to not only infiltration but also the advection and dispersion phenomena (Kamon et al., 2007). Therefore, these phenomena must be accurately reproduced in the implementation of the seepage and advection/dispersion analysis. In this study, the infiltration, advection and dispersion phenomena must be expressed three-dimensionally in order to account for the joint sections in the SPSP cutoff walls. Also, the analysis of coastal landfill sites, unlike that for inland landfill sites, must consider the effect of tides, etc. Furthermore, each vertical and bottom cutoff barrier is a composite structure consisting of synthetic fiber, steel, rubble and the seabed; this composite structure must be reproduced accurately.

The Eulerian-Lagrangean finite-element method is a numerical calculation method that is known to be useful in efficiently reproducing such complicated phenomena. In this study, the seepage and advection/dispersion analysis is performed using Dtransu-3D/EL, which is used as a representative analysis code (Nishigaki et al., 1995).

2.1 Objective and assessment index

In an SPSP cutoff wall, joint sections are arranged between steel pipes, forming a three-dimensional structure (see Fig. 2). Therefore, it is necessary to accurately reproduce the local leakage of leachates from the joint sections for the evaluation of the environmental feasibility of the SPSP cutoff wall. In this study, the leachate-containment effect of the SPSP cutoff wall is evaluated by using a three-dimensional seepage and advection/dispersion analysis (Dtransu-3D/EL). This analysis reproduces the existence of joint sections more precisely.

Figure 5 shows the three-dimensional cross-section of a landfill site assumed as a basic case in this analysis. The SPSP cutoff wall as well as a part of the composition layer around it in

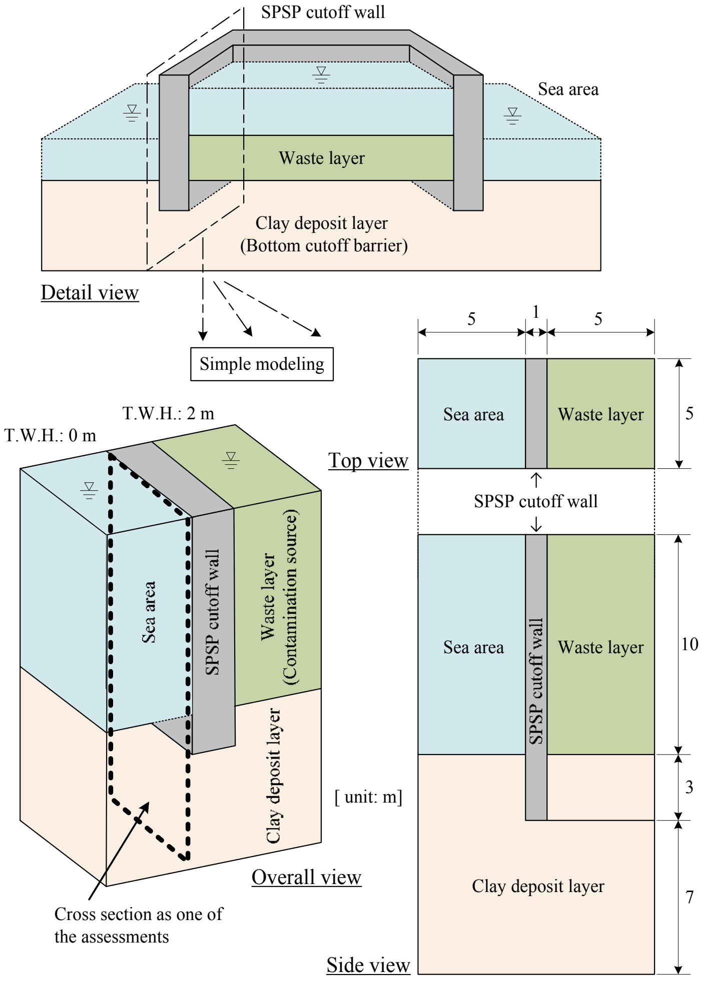

Fig. 5. Three-dimensional cross section of landfill site assumed as a basic case in the analysis

the coastal landfill site is considered for setting the three-dimensional cross-section. At the bottom of the waste layer as well as in the sea bed, a clay deposit layer is assumed to exist, and this layer fulfils the role as a bottom cutoff barrier in the coastal landfill site. The SPSP cutoff wall is penetrated upto a depth of 3 m in the clay deposit layer, and the hydraulic conductivity of the SPSP cutoff wall is varied to provide different examination cases.

In the construction of the SPSP cutoff wall at coastal landfill sites, double SPSP cutoff walls may be used due to ensure mechanical stability and fail-safe concept of landfill sites, as shown in the overview in Fig. 6. Furthermore, the clay deposit layer may be improved by sand compaction pile (SCP) methods in order to enhance the mechanical stability of the SPSP cutoff walls (Waterfront Vitalization and Environment Research Center, 2002). However, the main objective of this study is the evaluation of the environmental feasibility (containment effect of leachate containing toxic substances) of the SPSP cutoff wall. Therefore, the coastal landfill site is simplified, as shown in Fig. 5, as a three-dimensional cross-section that comprises a single SPSP cutoff wall, waste layer and clay deposit layer.

The three-dimensional cross-section assumes the extreme conditions for the vertical and bottom cutoff barriers that would pose environmental pollution risks to the surroundings affected by coastal landfill sites.

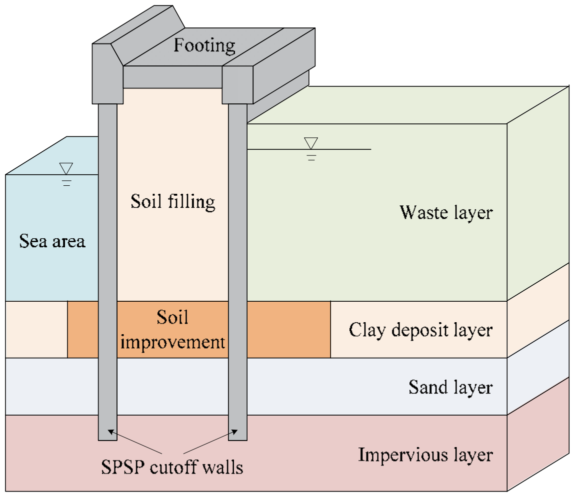

Fig. 6. Overview of vertical and bottom cutoff barriers generally constructing at coastal landfill sites

			SPSP cutoff wall			Clay deposit layer	Waste layer	Sea area
			UL-model	SP/JS-model				
				Joint sec.	Steel pipe			
Horizontal hydraulic conductivity	k_H	cm/s	2.0×10^{-6}, 1.0×10^{-6}, 1.0×10^{-7}, 1.0×10^{-8}	2.5×10^{-6}, 1.3×10^{-6}, 1.3×10^{-7}, 1.3×10^{-8}	infinitesimal	7.0×10^{-7}	1.0×10^{-0}	1.0×10^{-0}
Vertical hydraulic conductivity	k_V	cm/s	2.0×10^{-6}, 1.0×10^{-6}, 1.0×10^{-7}, 1.0×10^{-8}	2.5×10^{-6}, 1.3×10^{-6}, 1.3×10^{-7}, 1.3×10^{-8}	infinitesimal	5.0×10^{-7}	1.0×10^{-0}	1.0×10^{-0}
Effective porosity	θ		0.1	0.1	0.1	0.65	1	1
Longitudinal dispersion	α_L	cm	10	10	infinitesimal	10	10	10
Transverse dispersion	α_T	cm	0.1	0.1	infinitesimal	1	1	1
Molecule diffusion coefficient	D_m	cm^2/s	1.0×10^{-5}	1.0×10^{-5}	infinitesimal	1.0×10^{-5}	1.0×10^{-5}	1.0×10^{-5}
Retardation factor	R_d		1	1	1	2	1	1

Table 1. Seepage, advection and dispersion properties assigned to each composition layer in the analysis

In coastal landfill sites, the difference in the water level between the inside and outside landfill site is controlled on a daily basis so that it may not exceed 2 m (Waterfront Vitalization and Environment Research Center, 2002). On the other hand, in the three-dimensional cross-section shown in Fig. 5, a controlled water level regulated to 2 m is reproduced by the boundary conditions, that is, a fixed total head of 0 and 2 m are assigned to the upper sides of the sea area and waste layer, respectively. The boundary edges in the three-dimensional cross-section of the coastal landfill site are assumed to be undrained. The seepage, advection and dispersion properties assigned to each composition layer in this analysis are shown at Table 1. These values shown in Table 1 are typical one for heavy metals and composition layers (Kamon et al., 2001; Waterfront Vitalization and Environment Research Center, 2002). This analysis assumes that mechanical properties of each composition layer are not considered.

Presently, in Japan, waste discharge waste is burnt once at a refuse incinerator plant, and the incinerated residue generated from the incinerator plant is mainly used to reclaim land at landfill sites (Kamon & Inui, 2002). Therefore, the type of waste dumped in the recently constructed landfill sites has changed from the conventional organic substances to inorganic substances; thus, the heavy metals which may be contained in the incinerated residue are among the major environmental pollutants. If the leachate leakage occurs at a landfill site into the surrounding areas, the heavy metals also may leak out together with the leachate due to the advection-dispersion phenomenon, as heavy metals are soluble in water. Therefore, this study assumes heavy metals as toxic substances that may leak out from coastal landfill sites. This analysis assumes the waste layer to be a contamination source, and the concentration of toxic substances (heavy metals) at the waste layer is assigned the value of 1 as the initial condition. The initial relative concentration of toxic substances is initialized to 0 in regions outside the waste layer.

As an environmental conservation standard for coastal landfill sites (The Landfill System & Technologies Research Association of Japan, 2004), the environmental standard values (see Table 2 (b) and (c)) for water quality and bottom sediment of the sea areas near landfill sites equal 0.1 times that of the acceptable standard values (see Table 2(a)) for waste disposed at landfill sites. Therefore, the concentration of toxic substances at the SPSP cutoff wall on the sea side (that is the cross-section delimited by the broken line at Fig. 5) is targeted in this analysis as an important index of the environmental feasibility of SPSP cutoff walls. In this analysis, the elapsed time during which the concentration of toxic substances reaches 0.1 on the sea side of the SPSP cutoff wall is estimated; when this occurs, the SPSP cutoff wall as well as the coastal landfill site is defined as having lost its environmental feasibility.

2.2 SP/JS-model considering local water leakage in joint sections

In the evaluation of the environmental feasibility (containment effect of leachate containing toxic substances) of SPSP cutoff walls at coastal landfill sites, the equivalent hydraulic conductivity is generally used (Waterfront Vitalization and Environment Research Center, 2002). This method involves calculating the hydraulic conductivity of an SPSP cutoff wall equivalent to a uniform permeable layer of thickness 50 cm (see Fig. 4) by considering the steel pipes and joint sections that constitute the SPSP cutoff wall as a single body. Because the equivalent hydraulic conductivity can be directly verified with the technical standards for vertical and bottom cutoff barriers at landfill sites, it is frequently used in the technical development of the SPSP cutoff wall. However, the value equivalent hydraulic conductivity is the average hydraulic conductivity of the joint sections, which have high permeability, and that of the steel pipe sections, which are impermeable. Therefore, an evaluation using the equivalent hydraulic conductivity cannot easily detect the position or the time of leachate leakage, thus making it difficult to estimate the environmental impact of local leakage from the joint sections of the SPSP cutoff wall. Where, development of these detections will contribute strongly for the optimization of maintenance and management in SPSP cutoff wall.

Type of metals	Allowable limit
Cadmium and its compounds	0.1 mg/L or less
Lead and its compounds	0.1 mg/L or less
Hexavalent chromium compounds	0.5 mg/L or less
Mercury and its compounds	0.005 mg/L or less

(a) For industrial waste reclaimed in landfill sites

Type of metals	Allowable limit
Cadmium its compounds	0.01 mg/L or less
Lead and its compounds	0.01 mg/L or less
Hexavalent chromium compounds	0.05 mg/L or less
Mercury and its compounds	0.0005 mg/L or less

(b) For water pollution of groundwater

Type of metals	Allowable limit
Cadmium its compounds	0.01 mg/L or less
Lead and its compounds	0.01 mg/L or less
Hexavalent chromium compounds	0.05 mg/L or less
Mercury and its compounds	0.0005 mg/L or less

(c) For soil contamination

Table 2. Environmental conservation standards associated with inland and coastal landfill sites

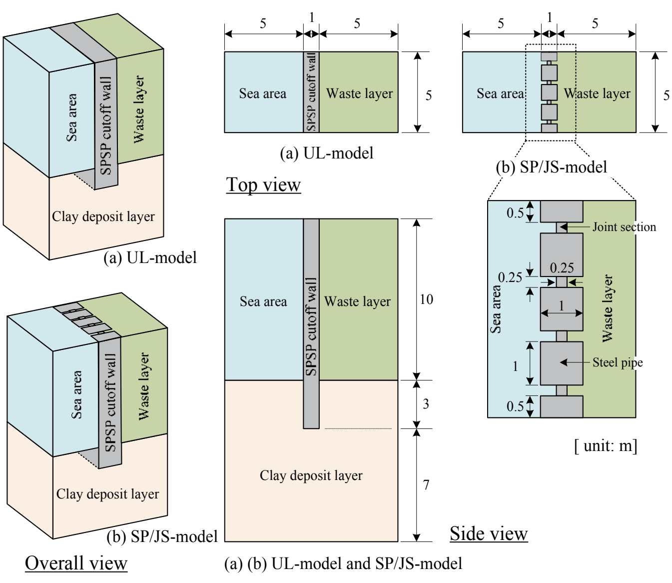

Fig. 7. General description of UL-model and SP/JS-model in the analysis

In this study, an evaluation method that can express the local leakage at the joint sections of SPSP cutoff walls is discussed. The evaluation method using the equivalent hydraulic conductivity is defined as the "UL-model", and the evaluation method that considers the steel pipe and joint sections, that is, the local leachate leakage, is defined as the "SP-JS-model". Figure 7 shows a general description of the UL-model and SP/JS-model. In the UL-model (shown in Fig. 7(a)), equivalent hydraulic conductivities of 2.0×10^{-6}, 1.0×10^{-6}, 1.0×10^{-7} and 1.0×10^{-8} cm/s were assigned to the entire SPSP cutoff wall. In the SP/JS-model (see Fig. 7(b)), the joint sections were placed at 0.25 m intervals for steel pipes of diameter 1 m, which represents the standard sizes of the SPSP shown in Fig. 2. Furthermore, hydraulic conductivities were assigned to each steel pipe and joint section in the SP/JS-model such that the entire hydraulic conductivity of the SPSP cutoff wall equals the equivalent hydraulic conductivity assigned in the UL-model, that is, hydraulic conductivities of 2.5×10^{-6}, 1.3×10^{-6}, 1.3×10^{-7} and 1.3×10^{-8} cm/s were assigned to the joint sections, assuming that the hydraulic conductivity of steel pipe is infinitely small. Table 1 shows the seepage, advection and dispersion properties assigned to each composition layer in both the models.

3. Results and discussion

3.1 Environmental feasibility of SPSP cutoff wall considering local water leakage

Figure 8 shows the concentration flux (the material quantity passing through a unit area in unit time) of toxic substances leaking from the SPSP cutoff wall on the sea side. The fluxes in the uniform layer of the UL-model and in each steel pipe and joint section of the SP/JS-model are plotted in Fig. 8. The relationship between the elapsed time and the highest concentration of toxic substances leaked from the SPSP cutoff wall on the sea side for both the models is shown in Fig. 9. Figure 10 illustrates the distribution of the concentration of toxic substances leaking out from the waste layer, which is the contaminated source, for both the models. Figure 10 expresses the distribution of the concentration on the sea side of the SPSP cutoff wall in order to facilitate the comparison of both the models with regard to the leakage of the toxic substance to the surroundings of the coastal landfill site.

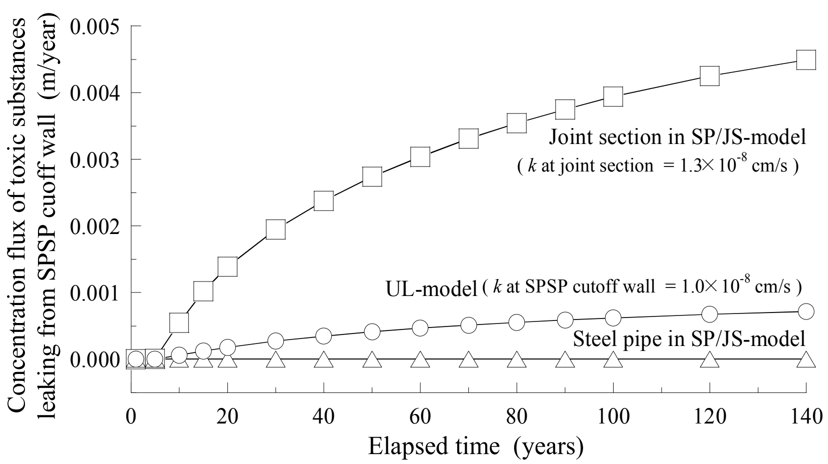

Fig. 8. Concentration flux of toxic substances leaking from SPSP cutoff wall on sea side with elapsed time for both models

In the SP/JS-model, the concentration flux of toxic substances leaked onto the sea side of the SPSP cutoff wall, particularly from the joint sections, is increased as compared to that of the UL-model (see Fig. 8). The SP/JS-model can quantitatively express the concentration of toxic substances at the joint sections of the SPSP cutoff wall, where the hydraulic conductivity is higher than that in the steel pipe. In the UL-model, as shown in Fig. 10, the leachate leaks uniformly from the SPSP cutoff wall onto the sea side, and this leakage tends to uniformly increase with time. In the SP/JS-model, it being different from the UL-model, the leachate leaks locally from the joint sections onto the sea side of the SPSP cutoff wall, and this leakage increases locally with time at the joint sections (see Fig. 10). Consequently, the increase in the concentration of toxic substances leaked from the SPSP cutoff wall onto the sea side is found to occur earlier in the SP/JS-model than in the UL-model, as shown in Fig. 9.

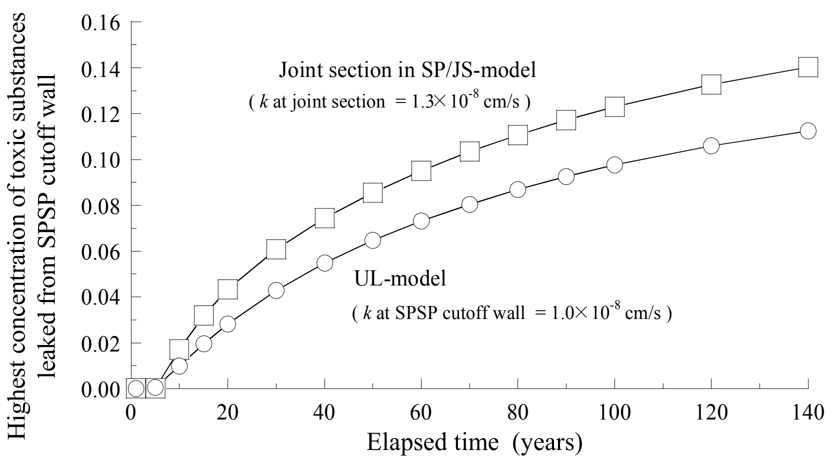

Fig. 9. Relationship between elapsed time and the highest concentration of toxic substances leaked from SPSP cutoff wall on sea side for both models

For example, 70 and 110 years, respectively, are required in the SP/JS-model (the hydraulic conductivity of the entire SPSP cutoff wall is 1.0×10^{-8} cm/s) and the UL-model for the concentration of toxic substances in the SPSP cutoff wall on the sea side to reach C=0.1, which is assumed as the assessment index. In the other analyzed conditions under which the hydraulic conductivity of the entire SPSP cutoff wall is equivalent in both models, the leakage of leachate is confirmed to occur earlier in the SP/JS-model than in the UL-model due to effect of the local leakage of leachate (see Fig. 11). This tendency becomes more remarkable with increase in the hydraulic conductivity of the entire SPSP cutoff wall (see Fig. 12).

Thus, as mentioned above, the reproduction of the local leakage of leachate generated at the joint sections of SPSP cutoff walls is possible by using the SP/JS-model for the evaluation of the environmental feasibility of SPSP cutoff walls at coastal landfill sites. Furthermore, the SP/JS-model indicates that toxic substances in concentrations exceeding the environmental standard values are leaked out of coastal landfill sites earlier than that estimated using the UL-model (see Fig. 9). Using the UL-model, the local leakage of leachate containing toxic substances from the SPSP cutoff wall cannot be reproduced, although the total quantity of the toxic substances leaked from the SPSP cutoff wall can be estimated. This provides a safer-side estimate of the environmental feasibility from the viewpoint of the time taken for the leakage of toxic substances. In addition, by using the UL-model, it is difficult to quantitatively detect the generation position in the SPSP cutoff wall where the leachate containing toxic substances are leaked. An appropriate estimation in terms of both position and time at which the loss of environmental feasibility occurs is important in order to control and maintain a long-term SPSP cutoff wall at coastal landfill sites. Based on the abovementioned points, the environmental feasibility of SPSP cutoff walls must be verified by using the SP/JS-model.

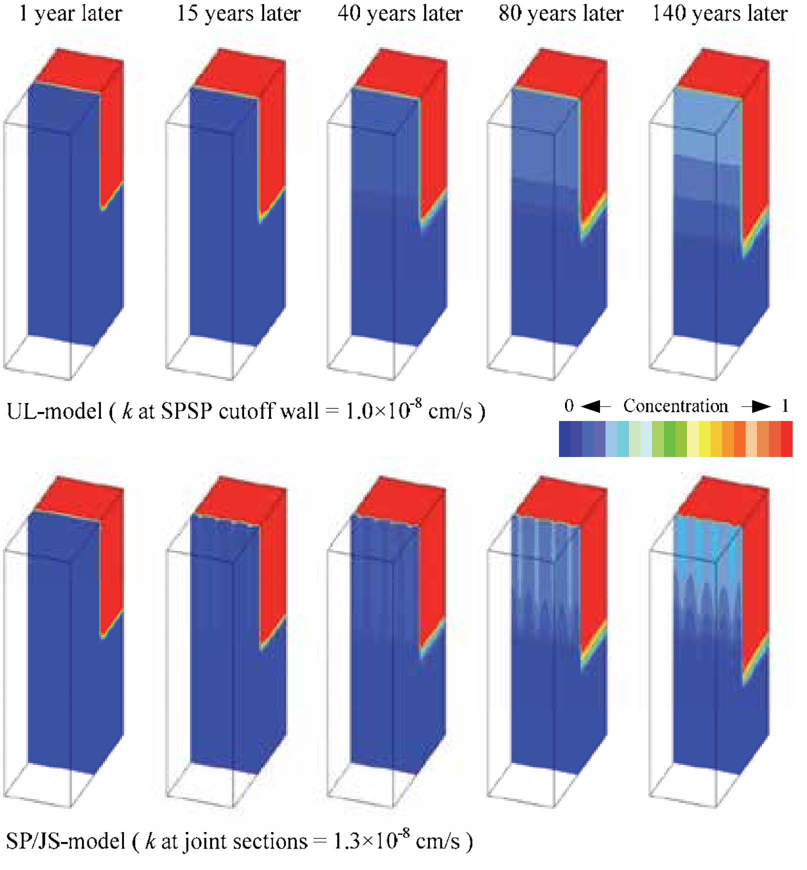

Fig. 10. Distribution of concentration of toxic substances leaking out from waste layer for both models

3.2 Environmental feasibility of SPSP cutoff wall considering joint sections

Various types of joints are adopted for the joint sections of the SPSP cutoff walls, as shown in Fig. 2. The types of joints for which the hydraulic performance has been reported experimentally are the P-T joints in which the packing mortar is filled in the joint space, the improved P-T joint in which a rubber board is installed with the mortar filling in the joint space and the H-H joint for H-jointed SPSP in which a water-swelling sheet is applied in the joint spaces (Oki et al., 2003; Inazumi et al., 2005, 2006; Kimura et al., 2007). Based on past reports, the SPSPs with the P-T joint, improved P-T joint and H-H joint exhibit equivalent hydraulic conductivity levels of 1×10^{-6}, 1×10^{-8} and 1×10^{-9} cm/s, respectively, under specific experimental conditions under which the difference between the water levels inside and outside the landfill site is less than 5 m (Oki et al., 2003; Inazumi et al., 2005, 2006). However, the reported hydraulic performances of the SPSP cutoff walls with the joint sections has been based on the equivalent hydraulic conductivities obtained from experimental studies.

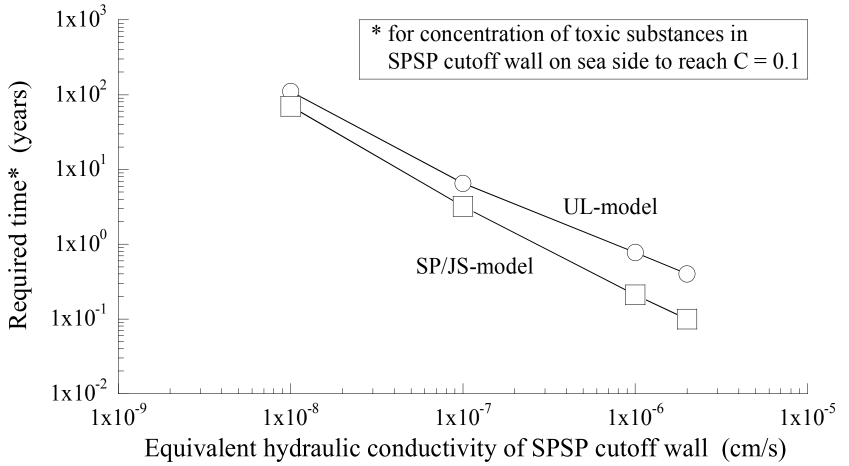

Fig. 11. Required time for concentration of toxic substances in SPSP cutoff wall on sea side to reach C = 0.1 with equivalent hydraulic conductivity of SPSP cutoff wall

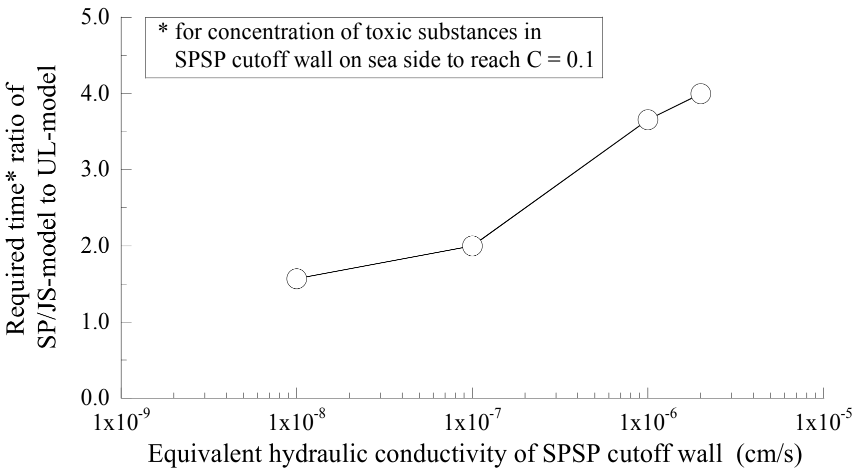

Fig. 12. Required time ratio of both models, for concentration of toxic substances in SPSP cutoff wall on sea side to reach C = 0.1, with equivalent hydraulic conductivity of SPSP cutoff wall

In this study, the reported equivalent hydraulic conductivities of SPSP cutoff walls are converted to individual hydraulic conductivities in the steel pipe and joint sections.

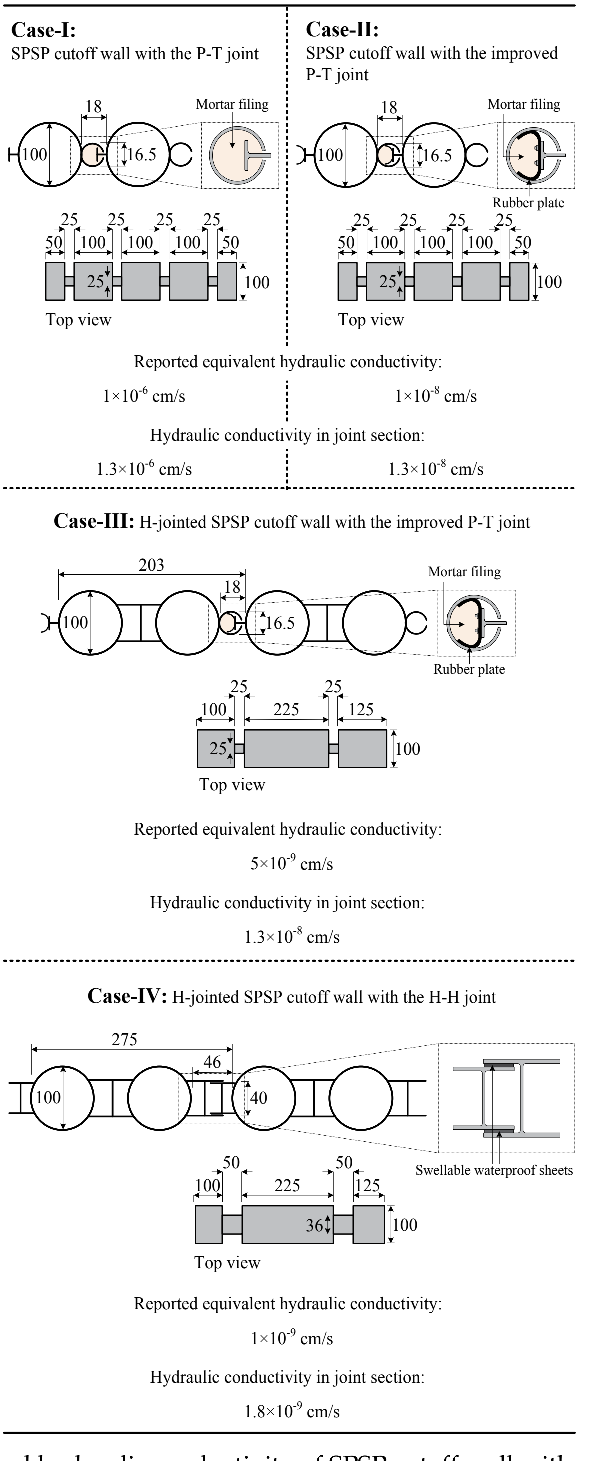

Fig. 13. Dimension and hydraulic conductivity of SPSP cutoff wall with each joint type and outline of SP/JS-model for Case-I to Case-IV

Furthermore, the environmental feasibilities of SPSP cutoff walls with various joints types are evaluated by applying each converted hydraulic conductivity in the SP/JS-model. Figure 13 shows the equivalent hydraulic conductivities of SPSP cutoff walls with various joints types, the dimension of each joint type as well as steel pipe and the hydraulic conductivity of each joint type. In the evaluation of the environmental feasibilities on SPSP cutoff walls considering various joint geometries and performance levels, the SPSP cutoff walls with the following four joint types are applied to the SP/JS-model.

Case-I: SPSP cutoff wall with the P-T joint
Case-II: SPSP cutoff wall with the improved P-T joint
Case-III: H-jointed SPSP cutoff wall with the improved P-T joint
Case-IV: H-jointed SPSP cutoff wall with the H-H joint

Figure 13 shows also the outline of the SP/JS-model for Case-I to Case-IV. Joint sections of width 0.25 m and steel pipes of diameter 1 m were used in Case-I and Case-II, whereas joint sections of widths 0.25 m and 0.5 m were used in Case-III and Case-IV, respectively, along with H-jointed steel pipes of diameter 2.25 m (Oki et al., 2003; Inazumi et al., 2005, 2006). Table 1 shows the seepage, advection and dispersion properties assigned to each composition layer

The assumed hydraulic conductivities of the joint sections were 1.3×10^{-6} cm/s in Case-I, 1.3×10^{-8} cm/s in Case-II and Case-III and 1.8×10^{-9} cm/s in Case-IV.

Figure 14 shows the total quantities of toxic substances leaked from the SPSP cutoff wall onto the sea side with respect to the elapsed time for Case-I to Case-IV. The relationships between the elapsed time and the highest concentration of toxic substances leaked from the SPSP cutoff wall onto the sea side for Case-I to Case-IV are shown in Fig. 15. Figure 16 shows the distribution of the concentration of toxic substances leaking out from the waste layer, that is, the contamination source, in Case-I to Case-IV. This distribution in Fig. 16 is expressed from the sea side of the SPSP cutoff wall in order to facilitate a comparison among Case-I to Case-IV with regard to the leakage of the toxic substance to outside the coastal landfill.

The times required for the concentration levels on the sea side to exceed C=0.1 were less than 1 year and 70 years for Case-I and the Case-II, respectively (see Fig. 15). In Case-III and Case-IV, the leakage of toxic substances in excess of environmental standard value (C=0.1) was not confirmed, even for durations upto 140 years. In Case-I and Case-II, the hydraulic conductivities of the joint sections are different, although the arrangement intervals of the joint sections are the same; thus it has been proven that low-hydraulic conductivity joint sections in SPSP cutoff walls significantly contribute toward increasing the leachate-containment effect. In addition, the sparser arrangement of joint sections represented in Case-III reduces the total quantity of toxic substances leaked from the SPSP cutoff wall onto the sea side to half that in Case-II (see Fig. 14). Consequently, the leachate leaked to the outside of the coastal landfill sites is reduced by the low hydraulic conductivity as well as the sparser arrangement of joint sections in the SPSP cutoff wall, thus, significantly improving the leachate-containment effect.

The H-jointed SPSP cutoff wall with H-H joints (Case-IV) most efficiently achieves low hydraulic conductivity with a sparser arrangement of joint sections. The leakage of leachates in Case-IV can be traced to the lower reaches of the cutoff wall, occurring via the clay deposit layer, which is one of the bottom cutoff barriers and is further away than other pathways such as leakage directly through the cutoff wall (see Fig. 16). Thus, the H-jointed SPSP cutoff wall with the H-H joint sufficiently contributes to the leachate-containment effect of vertical cutoff barrier at coastal landfill sites.

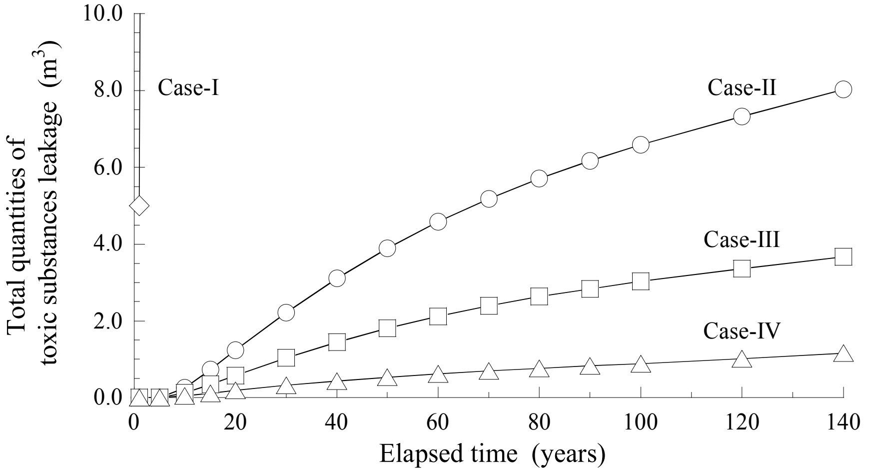

Fig. 14. Total quantities of toxic substances leaked from SPSP cutoff wall onto sea side with respect to the elapsed time for Case-I to Case-IV

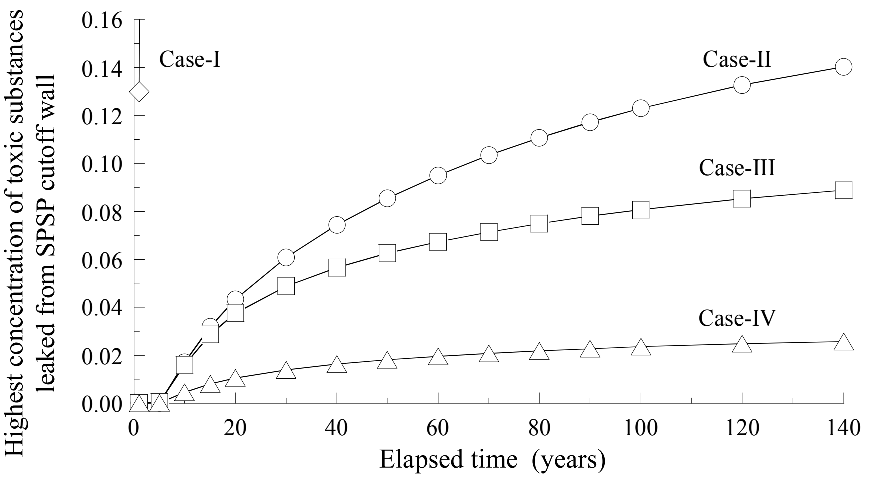

Fig. 15. Relationships between elapsed time and the highest concentration of toxic substances leaked from SPSP cutoff wall onto sea side for Case-I to Case-IV

In this section, it was clarified that technologies that lower the hydraulic conductivities of joint sections in SPSP cutoff walls and also facilitate the use of sparser arrangements contribute significantly to increasing the environmental feasibilities of SPSP cutoff walls at landfill sites. Further, the extent of the environmental feasibility of H-jointed SPSP cutoff walls with the H-H joints among the present technical developments in SPSP cutoff walls was shown.

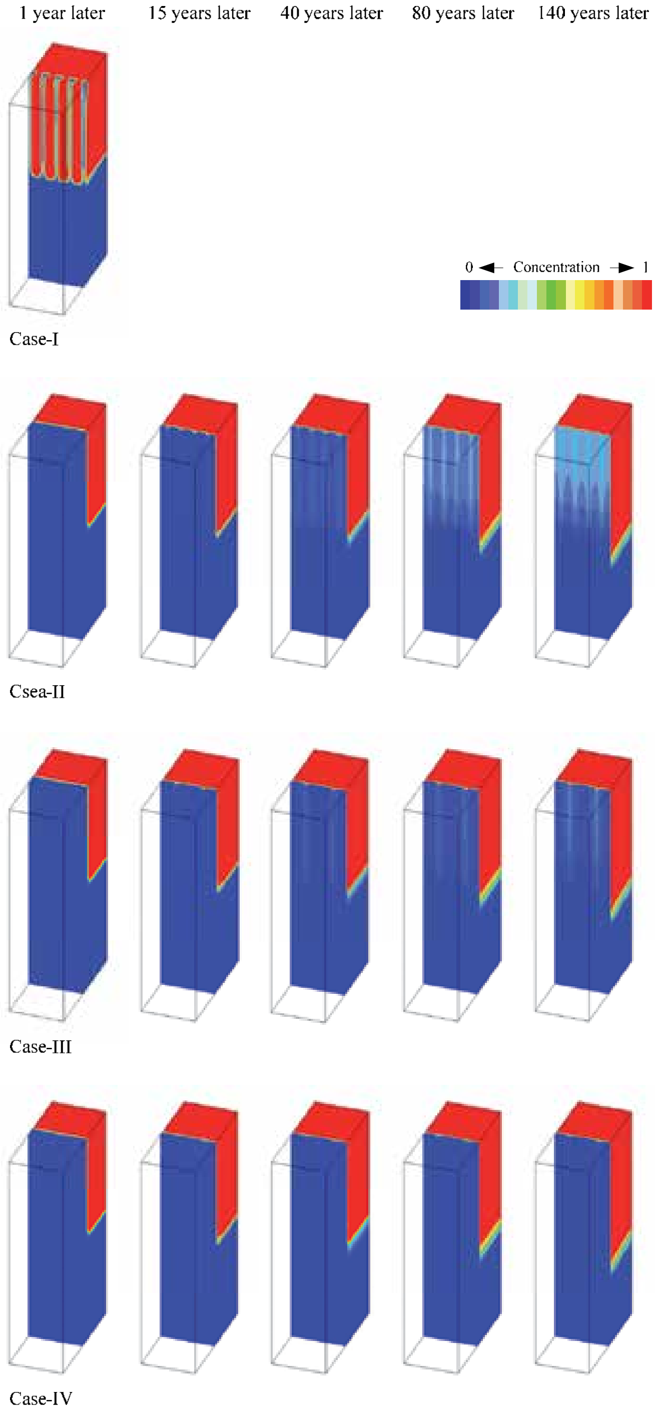

Fig. 16. Distribution of concentration of toxic substances leaking out from waste layer in Case-I to Case-IV

4. Conclusions

An evaluation method that can express the local leakage of leachate from joint sections in steel pipe sheet pile (SPSP) cutoff walls is discussed, in this study. In particular, the evaluation of environmental feasibility (containment of leachates containing toxic substances) considering a three-dimensional arrangement and hydraulic conductivity distribution of the joint sections in the SPSP cutoff wall is compared with an evaluation that generally uses the equivalent hydraulic conductivity.

Evaluations of the environmental feasibilities of the SPSP cutoff walls with joint sections that have a higher hydraulic conductivity than that of the steel pipe must take into account the local leakage of leachates containing toxic substances from the joint section; this was possible using the SP/JS-model. Due to the local leakage into the surroundings of coastal landfills from joint sections, contamination in excess of the environmental standard values was confirmed to occur earlier than that predicted by the UL-model, which is the current standard evaluation method.

It was clarified that technologies that lower the hydraulic conductivities of joint sections in SPSP cutoff walls and also facilitate the use of sparser arrangements contribute significantly to increasing the environmental feasibilities of SPSP cutoff walls at landfill sites. Further, the extent of the environmental feasibility of H-jointed SPSP cutoff walls with the H-H joints among the present technical developments in SPSP cutoff walls was shown.

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18

Environmental-Friendly Biodegradable Polymers and Composites

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France

1. Introduction

Global warming, the growing awareness of environmental and waste management issues, dwindling fossil resources, and rising oil prices: these are some of the reasons why "bio"products are increasingly being promoted for sustainable development.

"Bio"products, such as starchy and cellulosic polymers, have been used for thousands of years for food, furniture and clothing. But it is only in the past two decades that "bio"products have experienced a renaissance, with substantial commercial production. For example, many old processes have been reinvestigated, such as the chemical dehydration of ethanol to produce "green" ethylene and therefore "green" polyethylene, polyvinylchloride and other plastics. Moreover, recent technological breakthroughs have substantially improved the properties of some bio-based polymers, such as heat resistant polylactic acid, enabling a wider range of applications. In addition, plants are being optimized, especially to provide bio-fibres with more stable resource properties over time. An increasing number of applications have emerged recently (including packaging, biomedical products, textiles, agriculture, household use and building) where biodegradable polymers and biocomposites are particularly suitable as sustainable alternatives.

This chapter begins with a summary of the **classification systems for biodegradable polymers and biocomposites** then describes **specific and innovative developments concerning environmental-friendly biodegradable polymers and composites** carried out in recent years, based on several case studies:

- the development of a **multi-layered biocomposite based on expanded starch reinforced by natural fibres** for food packaging applications,
- the development of mulching and silage **films based on proteins extracted from cotton seeds** for agricultural applications,
- the development of a **biocomposite** for automobile applications **associating polylactic acid-based matrices and alterable glass fibres**,
- the formulation of **polylactic acid-based blowing films** for textile applications, such as disposable safety workwear,
- and the processing of **polylactic acid-based foam products** for several industrial sectors such as packaging and transport.

2. Classification systems

2.1 Classification of biodegradable polymers

Biopolymers can be classified in two ways: according to their renewability content (fully or partially bio-based or oil-based) and to their biodegradability level (fully or partially or not biodegradable) (Shen et al, 2009).

An attempt to classify biodegradable polymers into two main groups has been developed (Averous, 2004), these two groups being (i) the **agropolymers** obtained by biomass fragmentation processes (polysaccharides, proteins…), (ii) and the **biopolyesters** obtained either by synthesis from bio-derived monomers (polylactic acid – PLA) or by extraction from micro-organisms (polyhydroxyalkanoate – PHA) or by synthesis from synthetic monomers (polycaprolactone – PCL, aromatic and aliphatic copolyesters – PBAT, PBSA…) (Figure 1).

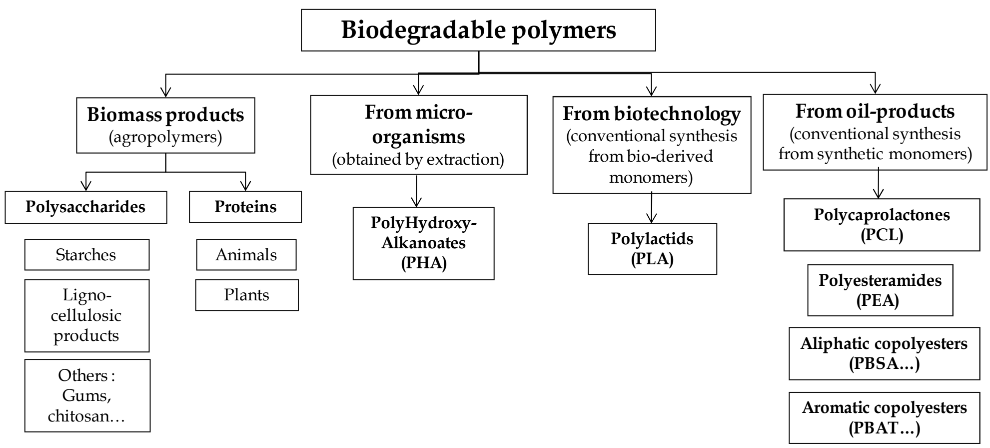


Fig. 1. Classification of biodegradable polymers (Averous, 2004)

2.2 Classification of biocomposites

The materials called biocomposites result from a combination of a biodegradable polymer and biodegradable fillers, usually bio-fibres.

Biocomposites can be classified into three main groups: (i) "**bio^{1}composites**", composites in which the production of raw materials is based on renewable resources, (ii) "**bio^{2}composites**" which are bio^{1}composites whose waste can be managed in an eco-friendly way at the end of their life (composting, biomethanation, recycling…), and (iii) "**bio^{3}composites**", which are bio^{2}composites where the successive transformation processes from the raw materials to the final products are environmental-friendly (low energy consumption, low emissions).

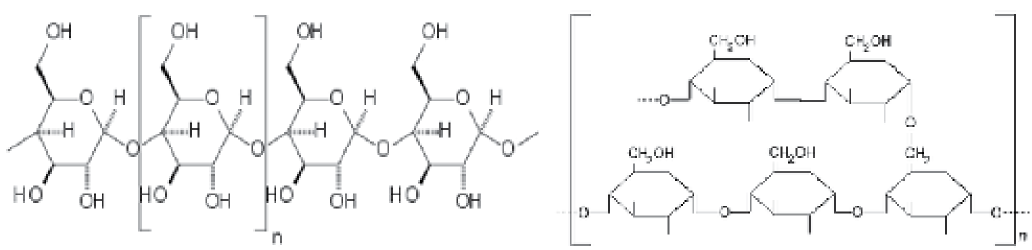
Nevertheless a problem remains: while it is relatively easy to define a "bio^{1}composite" by its content of renewable raw materials and a "bio^{2}composite" by its service-life/end-of-life time ratio, how can environmental efficiency be defined for "bio^{3}composite" transformation processes? With regard to the extrusion process, energy consumption can be evaluated from the specific mechanical energy (SME) and specific thermal energy (STE), which correspond respectively to the energy delivered by the screws per unit of mass of extruded biocomposite and to the total heat energy input through the barrel wall and the thermally regulated screws. A large number of energy efficiency indicators could be proposed for extrusion compounding such as the molten state viscosity of the extruded biocomposite and thermo-physical characteristics (transition temperature and enthalpy, heat capacity, thermal conductivity, density).

3. Agropolymer developments

Agropolymers include starch-based and protein-based polymers. After a general presentation of both types of polymers (microstructure, specific characteristics…) an example of innovative material development will be more extensively presented in each case.

3.1 Starch-based polymers and composites

Starch is the main storage supply in botanical sources such as cereals (wheat, maize, rice…), tubers (potato…) and legumes (pea…). In the past, studies carried on starch esters were abandoned due to their inadequate properties in comparison with cellulose derivates. It is only in the recent years that a renewed interest in starch-based polymers has been aroused.
Starch consists of two major components, amylose and amylopectine. Amylose (Figure 2a) is a linear or sparsely branched carbohydrate based on α(1-4) bonds with a molecular weight of 10^5-10^6. The chains show spiral shaped single or double helixes. Amylopectine (Figure 2b) is a highly multiple branched polymer with a high molecular weight of 10^7-10^9 based on α(1-4) bonds and α(1-6) links constituted branching points occurring every 22-70 glucose units (Zobel, 1988; Averous, 2004). In nature starch is found as crystalline beads, in three crystalline modifications according to the botanical source.



(a) (b)

Fig. 2. Structures of (a) amylose and (b) amylopectine

Apart from its use as a filler to produce reinforced polymers (Griffin, 1973), most starch applications require water and the disruption of the granular structure, which is called gelatinization. Starch can swell to form a viscous paste with most of its inter-macromolecule hydrogen bonds being destroyed. A reduction in both melting and glass transition temperatures is observed. It can be shown (Averous, 2004) that different products are obtained in function of the level of destructuring and the water content.
It is for that reason that starchy materials are divided into two categories: (i) with a high water content (between 15 and 30% in volume), **expanded starches** are obtained by expanding starch in the presence of specific blowing and nucleating agents through an extrusion die; (ii) with a low water content (below 15% in volume), **plasticized starches**, also called “thermoplastic starches” (TPS), are obtained after disruption and plasticization of the starch by applying thermo-mechanical energy in a continuous extrusion process.

Starchy materials present some drawbacks compared to conventional oil-based polymers such as a strongly hydrophilic character and rather poor mechanical properties. These weaknesses could be improved by blending with less water sensitive biopolymers and incorporating cellulose-based fibres.

3.1.1 Biocomposites based on plasticized starch

Plasticized starches have been combined with various fibres such as jute fibres (Soykeabkaew et al, 2004), ramie fibres (Wollerdorfer & Bader, 1998), flax fibres (Soykeabkaew et al, 2004; Wollerdorfer & Bader, 1998), tunicin whiskers (Angles & Dufresne, 2001), bleached leaf wood fibres (Averous et al, 2001), wood pulp (De Carvalho et al, 2002) and microfibrils from potato pulp (Dufresne et al, 2000). Most of these authors have shown a high compatibility between starch and cellulose-based fibres leading to higher moduli. A reduction in water sensitivity is also obtained because of the more hydrophobic character of cellulose, which is linked to its high crystallinity. Another reason for the improved properties of fibre reinforced starch biocomposites is the formation of a tight three-dimensional network between the carbohydrates through hydrogen bonds.

3.1.2 Biocomposites based on expanded starch: development of a multi-layered biocomposite for food packaging applications

The materials used for packaging today consist of a variety of petroleum-derived polymers (mainly polyolefin such as polyethylene, polypropylene and polystyrene), metals, glass, paper and combinations thereof. Concerning food products, they must have specific optimum requirements especially regarding storage and interaction with food. The engineering of new bio-based food packaging materials can thus be considered as a tremendous challenge both for academia and industry.

Our research centre and Vitembal Co (Remoulins, France) have joined forces to develop an innovative multi-layered biodegradable composite intended to replace the common Expanded PolyStyrene (EPS) trays used for food packaging, especially fish, meat and vegetables. Starch was considered as a suitable alternative for achieving the required foamed structure. The project was supported by the French organization ADEME.

3.1.2.1 The multi-layer concept

The starch (potato starch provided by Roquette Co, France, with 10-25 wt% amylose, 75-80 wt% amylopectine, 0.05 wt% proteins based on dry weight) used for this study was expanded through a classical co-rotating extruder (Clextral BC21, 900 mm length, 25 mm diameter, 1.5x40 mm^2 flat die) with 12 heating zones (temperature profile: 30°C (feeder) / 30°C / 50°C / 60°C / 70°C / 80°C / 90°C / 90°C / 100°C / 120°C / 120°C / 160°C (die)) to obtain sheets that were afterwards thermoformed to shape the final tray.

The expansion was induced by water added using a peristaltic pump. An optimized value of 17 wt% of water was obtained, leading to the best expansion. Regular expansion was achieved by adding 2 wt% of talc (Talc de Luzenac Co, France) and 2 wt% of a chemical blowing agent (CBA) based on citric acid and sodium bicarbonate (Hydrocerol ESC5313© supplied by Clariant Co, France). It can be noticed that the foaming aptitude of starch was assessed on the basis of void content induced by extrusion in the final product. The experimental results enabled the definition of an optimum set of extrusion conditions (screw profile and speed, cooling temperature, extrusion temperatures along the screw…) and

material formulations (CBA content, viscosity of polymer during processing…), leading to a maximal void content.
Nevertheless, the main drawbacks of starch are its high water sensitivity and poor mechanical properties. Therefore, firstly, natural fibres were incorporated within the starch. Various natural fibres such as wheat straw fibres, cotton linter fibres, hemp fibres and cellulose fibres (Table 1) and fibre contents (7, 10 and 15 wt %) were compared. In addition, two external biodegradable low hydrophilic polyester films (120 μm) of polycaprolactone (PCL) were calendared on both sides of the core sheet of foamed starch, to limit water absorption and enhance global mechanical properties.
Under these conditions, all the formulations were processed with specific mechanical energy (SME) values between 60 and 90 W.h/kg. The final multi-layered biocomposite structure is presented in Figure 3.

Fibre	Length (mm)	Cellulose content (%)	Supplier
Wheat straw	2.6	30-35	A.R.D. Co (France)
Cotton linter	2.1	80-85	Maeda Co (Brazil)
Hemp	3.2	70-72	Chanvrière de l'Aube (France)
Cellulose	0.13	98-99	Rettenmaier and Söhne (Germany)

Table 1. Main characteristics for different natural fibres used

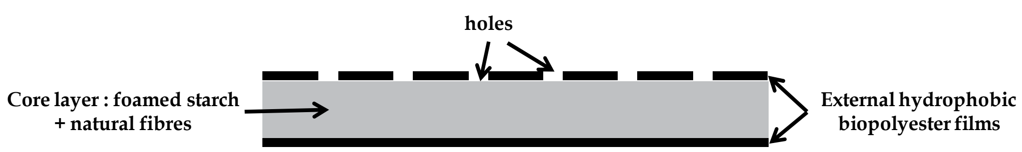


Fig. 3. Multi-layered biocomposite structure

3.1.2.2 Properties of the biocomposite core layer

3.1.2.2.1 Density, expansion index and cell morphology

It is noticeable (Table 2) that the addition of fibres contributed to lowering the core layer density except in the presence of hemp fibres. A slight reduction in expansion index was observed in the presence of cellulose and hemp fibres, whereas an increase was observed in the presence of wheat straw and cotton linter fibres. These effects may result from two competitive mechanisms varying according to the nature of the fibre: on the one hand fibres tend to increase the viscosity of the moulded starch but, on the other hand, fibres act as nucleating agents providing surfaces for cell growth.
As a consequence, reinforced starch foams exhibit smaller cells (mean diameter between 580 and 780 μm compared to 880 μm for unreinforced foamed starch) with thinner walls (between 12.5 μm and 18.6 μm compared to 21.5 μm for unreinforced foamed starch) as shown in Table 3 and Figures 4a to 4c. The results show an open-cell structure (around 80% of open-cells) for all formulations, with little variation between the various formulations, this parameter being mainly influenced by processing conditions and especially cooling speed at the extruder die. The microstructure of industrial multi-layered EPS trays is very different. Indeed, industrial EPS trays are a two-layered system with an open-cell layer (75-85% of open-cells) in contact with the food for optimized absorption of exudates and a closed-cell layer (85-95% of closed cells) to act as a diffusion barrier. Moreover EPS cells are

smaller (about 300 μm) (Figures 4d and 4e). As a consequence it can be concluded that the main challenge was to control the microstructure of the starch foam (i.e. rate of open-cells, cell size, wall thickness).

Fibre	Content (wt%)	Density (g/cm^3)	Expansion index
Wheat straw	7	0.225 ± 0.021	3.0 ± 0.2
	10	0.190 ± 0.012	3.5 ± 0.1
	15	0.186 ± 0.009	3.2 ± 0.2
Cotton linter	10	0.175 ± 0.008	3.2 ± 0.1
Hemp	10	0.242 ± 0.010	2.8 ± 0.2
Cellulose	7	0.161 ± 0.004	2.9 ± 0.1
	10	0.170 ± 0.007	2.8 ± 0.1
	15	0.158 ± 0.004	2.4 ± 0.1

Table 2. Densities and expansion ratios of starch based biocomposites compared to starch (density: 0.236 ± 0.016; expansion index: 2.9 ± 0.2)

	D_n (μm)	D_w (μm)	PDI	e (μm)	I_s
Wheat straw	653.8	812.7	0.80	18.61	0.70
Cotton linter	648.9	734.1	0.88	15.12	0.70
Hemp	784.1	966.9	0.81	17.39	0.70
Cellulose	577.6	730.7	0.79	12.54	0.70

Table 3. Size (mean diameter in number, D_n;. mean diameter in weight, D_w), wall thickness of cells (e), polydispersity index (PDI) and sphericity (I_s) of biocomposites reinforced by 10 wt% of fibres compared to starch (D_n: 875.5 μm; D_w: 1046.6; PDI: 0.84; e: 21.52 μm; I_s: 0.72)

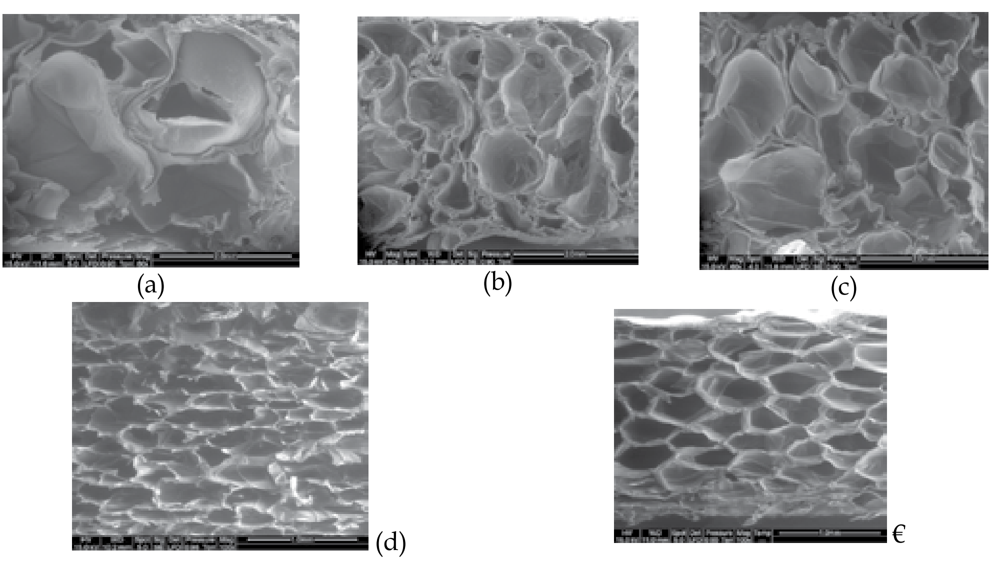


Fig. 4. Cell morphology (a) starch; starch biocomposites reinforced by 10 wt% of (b) wheat straw; (c) cellulose fibres; (d) open- and (e) closed-cells structure of an EPS tray

3.1.2.2.2 Water absorption

Water absorption was measured after storing samples at various relative moieties (33, 56 and 75 RH %) for 200h. This water sensitivity was measured both for the fibres alone (Table 4) and for the core layer of the biocomposites (Table 5).
It was observed that the water absorption of the fibres was lower than that of foamed starch under the same conditions (9.1; 12.5 and 16.8 % respectively for 33; 56 and 75 RH %). It would therefore be expected that the presence of fibres would lower the water sensitivity of the expanded starch. However, this decrease in water absorption seems to depend on the type of fibre. Cotton linter fibres show much lower water sensitivity than the other fibres, but such a difference is not observed for the corresponding biocomposite. Two main explanations could be proposed, the first concerns the influence of cell morphology, especially wall thickness, on water vapour diffusion within the material, and The second concerns the potential existence of interactions between fibres and matrix through hydrogen bonds that modify water-fibre and water-starch interactions.

Fibres	33 RH %	56 RH %	75 % RH
Wheat straw	4.5	7.6	11.5
Cotton linter	3.8	6.1	9.0
Hemp	5.0	7.8	11.8
Cellulose	5.3	7.7	11.5

Table 4. Water absorption rate of isolated natural fibres at various relative moieties for 200h

	Wheat straw			Cotton linter	Hemp	Cellulose		
	7 %	10 %	15 %	10 %	10 %	7 %	10 %	15 %
33 %	8.7	8.4	8.3	9.5	8.8	9.2	8.8	9.1
56 %	11.9	11.7	11.4	12.4	12.0	12.5	11.5	12.2
75 %	16.0	15.6	15.3	16.2	16.4	16.7	15.7	15.9

Table 5. Water absorption rate of the core layer of the biocomposites with different weight contents of fibres (7, 10, 15 wt%) and various relative moieties (33, 56, 75 %RH) for 200h

3.1.2.2.3 Mechanical properties

Equivalent E/ρ values (E: bending modulus; ρ: density) for fibre reinforced biocomposites (10 wt% of fibres) are presented in Figure 5 for all humidity rates. It can be observed that cellulose fibres confer the most significant reinforcement effect to the starch foam, followed by hemp and linter cotton fibres. Moreover an increase in relative humidity level results in a decrease in the mechanical properties. This is related to the plasticizing effect of water with respect to starch. Despite the fact that natural fibres are less water sensitive than starch, it is observed that the incorporation of fibres in starch foam does not systematically lead to a reduction in hygroscopicity and thus an improvement in mechanical properties.

3.1.2.2.4 Biodegradation rate

Different degradation tests were investigated on the core layer of the different developed biocomposites.
The weight variation of the biocomposite versus composting time was measured (composting test – ISO 14855) (Table 6). The presence of fibres may delay the degradation

rate for short composting times, but a degradation rate of between 38 and 51% was obtained after 4 months whatever the fibre nature due to fungal growth (Aspergillus, Hyphomycetes).
The oxygen consumption of micro-organisms (BOD: Biological Oxygen Demand - ISO 14432) shows a lower degradation rate after 28 days for biocomposites compared to unreinforced foamed starch (Table 6). This could be explained by the fact that starch degradation may occur before fibre degradation. The activated sludge issued from a wastewater treatment may contain bacteria that can more easily produce enzymes for starch degradation than for fibre degradation. The variations in BOD according to the nature of the fibres may be due to an acclimation period of 28 days for fibre degradation.

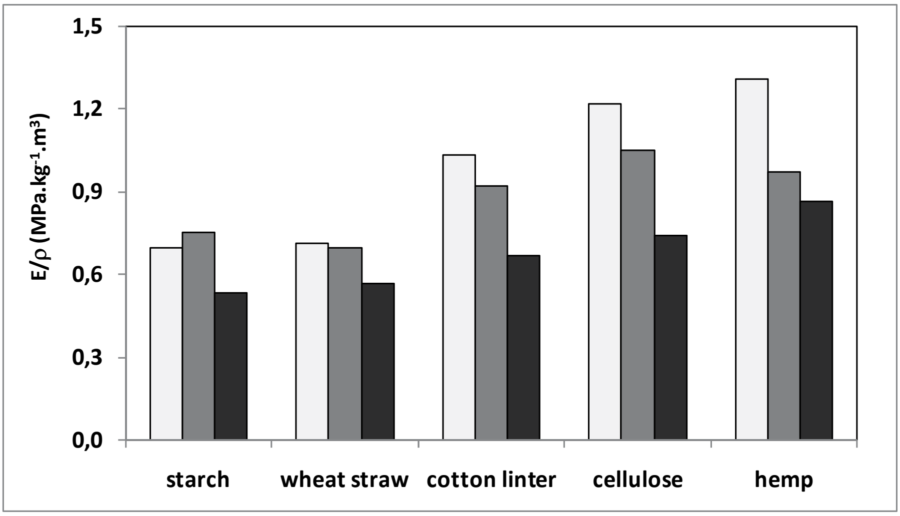


Fig. 5. E/ρ of unreinforced starch and of starch based biocomposite (10 wt% of fibres) as a function of relative humidity (□ : 33 HR%; ▩ : 56 RH%; ■ : 75 RH%) (E: bending modulus; ρ: density)

	Composting time	Foamed starch	+10 wt% wheat straw	+10 wt% cotton linter	+10 wt% hemp	+10 wt% cellulose
ISO 14855	32 days	42.7	27.6	30.9	29.2	27.4
	53 days	42.8	32.1	31.4	29.4	25.5
	88 days	41.6	30.3	29.1	28.8	30.8
	122 days	50.9	47.3	43.6	37.8	49.7
ISO 14432	28 days	73	51	52	66	67

Table 6. Degradation rate (%) of the core layer of various biocomposites with composting time according to ISO 14855 and ISO 14432

3.1.2.3 Properties of the multi-layered biocomposite

As concerns the multi-layered biocomposite (Figure 3), results show an increase in density and mechanical properties compared to the core layer alone. Higher impact strengths and similar E/ρ values were obtained for starch-based biocomposites than for EPS (Figure 6). Nevertheless water sensitivity remained ten times lower for the biocomposite (absorption rate about 1 g/dm^2 whatever the fibre nature after 24h in contact with a physiological serum) by comparison to EPS (12 g/dm^2 under the same conditions). At the same time a drastic decrease in mechanical properties was observed. The oxygen consumption of microorganisms (BOD) shows a lower degradation rate for the multi-layered systems compared to the core layer alone, with values between 23 and 28% instead of 51-67%.

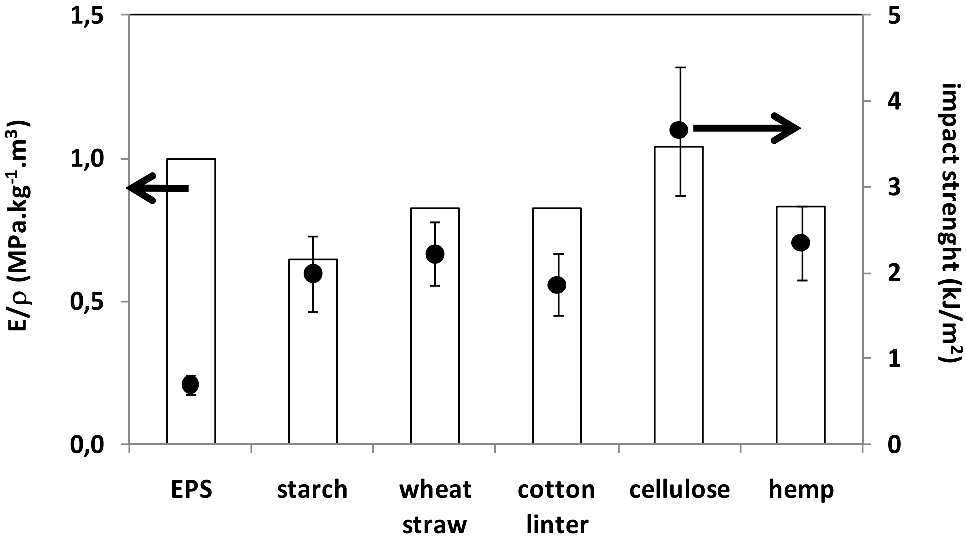


Fig. 6. E/ρ values (E: bending modulus; ρ: density) and impact strengths of the various multi-layered biocomposite (10 wt% of fibres) compared to EPS (commercial tray) at 56 RH %

3.1.3 Further studies

Current studies are focussing on four main topics: (i) optimisation of the cell morphology to reduce the cell size through the incorporation of nanofillers, (ii) control of the open/closed-cells structure through optimisation of the processing conditions, (iii) the use of other natural fibres to modulate the mechanical properties and (iv) the appliance of specific surface treatments on the natural fibres to reduce the water sensitivity of the biocomposite and increase fibre/starch interactions.

3.2 Protein-based polymers and composites: development of mulching and silage films for agricultural applications

3.2.1 General aspects

A wide range of materials have been successfully prepared from proteins, which are abundant and inexpensive. It is well known that the mechanical properties of protein-based materials correlate with the density of the three-dimensional network formed during processing through disulfide-bond crosslinking (Domenek et al, 2002; Shewry & Tatham, 1997). This density increases with the processing temperature and duration, resulting in higher tensile strength and Young's modulus while elongation at break decreases (Morel et al, 2002). Nevertheless optimal processing conditions need to be defined for which thermal aggregation is maximized while the degradation mechanism is still negligible. Plasticizers, as well as natural fibres, may modify both the processing window and mechanical properties.

The engineering of protein-based biodegradable polymers is therefore providing challenging alternatives for agricultural items, like mulching films, silage films, bags and plant pots. With a worldwide production of about 33 million metric tons, cottonseed cakes are now the most important source of plant proteins after soybeans. These products seem to be very attractive for non alimentary applications such as developing a biodegradable polymer. Nevertheless, in most cases, wet processes such as casting are used for these materials. The objective is to use the dry processing technologies (extrusion, thermo-moulding) currently used for synthetic polymers.

3.2.2 Protein-based films obtained through dry technologies

Our research centre was involved in a FP5 European project to develop protein-based biopolymers through dry processes. This research program was managed by the CIRAD (Centre de Coopération Internationale en Recherche Agronomique pour le Développement, Montpellier, France) and was carried out in collaboration with South American companies and institutions (Brazil, Argentina).

Dry technologies imply that proteins exhibit thermoplastic behaviour, i.e. a viscous flow at high temperature. In many cases, the glass transition of proteins occurs very close to the temperature of thermal degradation. To enlarge the processing range, proteins are mixed with small molecules intended to lower the glass transition temperature by plasticization. Due to the hydrophilic nature of many amino acids, polyols (glycerol, sorbitol…) are commonly used for protein plasticization.

The influence of several parameters was investigated: plasticizer nature and content, storage conditions, presence of shells, presence of lipids, processing conditions. Results highlighted that the presence of plasticizers tends to decrease Young's modulus and tensile strength and to increase elongation at break. This effect increased with plasticizer content and the number of hydroxyl groups supplied by the plasticizer. Storage conditions also have a major influence on mechanical properties, water being a good plasticizer of proteins. The presence of shells tends to reduce the mechanical performance of the films. At very low content (<2 wt %), shells can promote a positive effect by increasing the tensile strength and rigidity. Above 2 wt %, shells decrease the mechanical strength because they act as crack initiators due to their morphology and poor adhesion to the protein matrix. The presence of lipids decreases the rigidity of the materials, with poor cohesion. This was attributed to phase separation between the lipids and proteins. Finally, concerning the influence of processing conditions, the best results were obtained when films were pressurised at 120°C. At lower temperatures, the cohesion of the films was poor (low Young's modulus). At higher temperatures, elongation at break decreased due to potential crosslinking reactions or degradation reactions. The tensile strength =f(elongation at break) curve (Figure 7) shows that the best results were obtained when the films were plasticized with glycerol, were processed at 120°C and contained a small amount of shells.

4. Developments regarding biopolyesters

Biopolyesters are obtained (i) **from biotechnology** (conventional synthesis from bio-derived monomers) such as polylactides (PLA), (ii) **by extraction from micro-organisms** such as polyhydroxyalkanoates (PHA) and (iii) **from petrochemical products** (conventional synthesis from synthetic monomers) such as polycaprolactones (PCL) and aromatic and aliphatic copolyesters. A wide range of these biodegradable polymers is now commercially available, offering all sorts of properties that enable them to compete with non-biodegradable polymers in several industrial sectors.

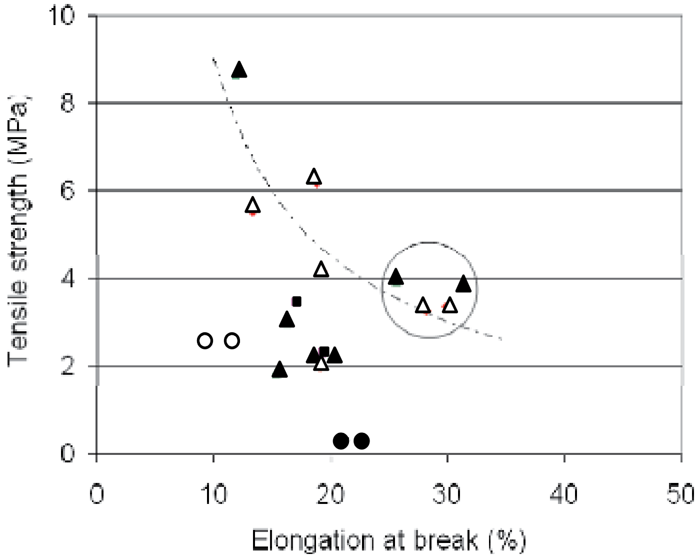


Fig. 7. Tensile strength as a function of elongation at break for various protein-based films prepared under different conditions : plasticized by glycerol and processed at ● 90°C, ∆ 120°C, O 140°C, ▲ 120°C containing shells and ■ plasticized by triethanolamine and processed at 120°C

4.1 Polylactic acid-based polymers

PLA is currently one of the most promising biopolymers. During the last decade PLA has been the subject of an abundant literature, with several reviews and book chapters (Averous, 2004; Garlotta, 2002; Auras et al, 2004; Mehta et al, 2005; Sodergard & Stolt, 2002). Processable by many techniques (blowing films, injection moulded pieces, calendared and thermoformed films…) a wide range of PLA grades is now commercially available with companies such as Cargill (USA), Mitsui Chemical (Japan), Galactic (Belgium), Shimadzu Co (Japan), Purac (Netherland) and many others (Shen et al, 2009).

After a general presentation of the synthesis and properties of polylactic acid, this chapter will detail three case studies.

The first case study concerns a **biocomposite for automobile applications combining a PLA-based matrix and an alterable glass fibre**. The challenge of the work is twofold: maintaining constant mechanical properties under aggressive conditions (temperature, moisture, mechanical stresses) during the in-service life of the automobile and having a material that is easily recycled by composting at the end of its life.

The second study case concerns **PLA-based films for some textile applications**, especially disposable safety workwear. With identical performances to non woven tissues or polyolefin weldable films, PLA films are considered to be competitive alternatives because of their

biodegradability. The required performances (tear resistance, weldability, perforation, thermal resistance, barrier properties…) can be achieved by incorporating specific additives (plasticizers, chain extender molecules, crosslinking agents…) and defined nanoparticles.
The third study case concerns **PLA-based foam products**. With the aim of reducing the environmental impact of plastics, these materials are of major industrial interest, replacing heavy items by lighter bio-based products with identical performance levels. They could be considered as interesting alternative candidates to polyethylene foams, for example, with expansion rates of about 50%**.** The objective of the studies concerned is to optimize either the processing conditions (extrusion flow rate, temperature, cooling system) or the material formulation (content of chemical blowing agent, PLA characteristics) for maximum foam expansion and good mechanical performances.
Finally it is important to underline that PLA is considered as one of the three biodegradable polymers used for **clinical applications**, together with polyglycolic acid (PGA) and paradioxanone (PDS). Copolymers of PLA and PGA remain the most interesting alternatives to metals for bone consolidation. These applications will not be detailed in this chapter.

4.1.1 Synthesis and properties of PLA

Lactic acid is extracted from starch and converted to a high molecular weight polymer (Mw>100000) through an indirect polymerization route via lactide. This route was first demonstrated by Carothers in 1932 (Carothers, 1932) but high molecular weights were not obtained until improved purification techniques were developed (Garlotta, 2002). The mechanism involved is ring-opening polymerization (ROP) and may be ionic or coordination-insertion depending on the catalytic system used (Auras et al, 2004; Sodergard & Stolt, 2002; Stridsberg et al, 2001; Mehta et al, 2005).
All properties of PLA depend on its molecular characteristics, as well as the presence of ordered structures (crystalline thickness, crystallinity, spherulite size, morphology and degree of chain orientation). The physical properties of polylactide are related to the enantiomeric purity of the lactic acid stereo-copolymers. PLA can be produced totally amorphous or up to 40 % crystalline. PLA resins containing more than 93 % of L-lactic acid are semi-crystalline, while those containing 50–93 % are entirely amorphous. The typical PLA glass transition temperature ranges from 50°C to 80°C, whereas the melting temperature ranges from 130°C to 180°C. The mechanical properties of PLA can vary considerably, ranging from soft elastic materials to stiff high strength materials, according to various parameters, such as crystallinity, polymer structure, molecular weight, material formulation (plasticizers, blend, composites…) and processing. For instance, commercial PLLA (92% L-lactide) has a modulus of 2.1 GPa and an elongation at break of 9 %. The CO_2 permeability coefficients for PLA polymers are lower than those reported for crystalline polystyrene at 25°C and 0 % of relative humidity and higher than those for PET. The main abiotic degradation phenomena of PLA involve thermal and hydrolysis degradations.

4.1.2 Polylactic acid-based biocomposites for automobile applications

It is well known that the development of automobile parts requires materials with high mechanical characteristics and good thermal properties that remain constant throughout the in-service life of the automobile in a potential aggressive environment. This challenge could be achieved by the incorporation of reinforcements. Natural fibres are commonly used to reinforce PLA because of their renewability and biodegradability. Moreover, their low price

and low density are complementary advantages. Unfortunately, their main drawbacks are their relative low mechanical properties depending on the production location and crop, the weakness of the matrix/fibre interface and the potential competition with food production.
Our research centre and OCV Chambéry International (Chambéry, France) joined forces to develop an innovative biodegradable biocomposite reinforced by glass fibres that can be degraded by water and mineralized by microorganisms without any toxic components being released into the environment. This alterability, combined with the good reproducibility of glass fibre properties compared to plant fibres, shows considerable promise. This research program was supported by the French organizations ADEME and ANR.

4.1.2.1 Alterable glass fibres

Most of the alterable glasses developed in recent years have been used in medical applications. They are based on silicate, calcium and phosphate components, leading to an improvement in micro-organism activity. They are incorporated within several biodegradable polymers such as PLA (Zhang et al, 2004) allowing bone reconstruction. Several steps are involved in glass alteration: inter-diffusion (exchanges between alkaline components of the glass and the solution), glass hydrolysis (direct interaction with intrinsic glass network), gel formation (re-condensation of some components such as silicates), precipitation of secondary phases. Several parameters may influence these mechanisms, such as glass composition, pH, temperature, contact surface and micro-organisms. For this project the main alterable glass formulations were based on silicate and moreover present the advantage of having a lower melting temperature and therefore lower energy requirements for processing than conventional glasses.

4.1.2.2 Mechanical properties of biocomposites

An alterable glass fibre (AGF) and a conventional E-glass fibre (E) were compared, both coated with a standard water based sizing used in traditional polyester composites (such as PET or PBT). A comparison with hemp natural fibre (HNF) will be also carried out. Classical co-rotating extrusion (Clextral BC21) and injection molding (Sandretto) processes were used for the elaboration of PLA (PLA 7000D© provided by Nature Works LCC, USA) reinforced by 30 wt% of fibres.

		PLA	PLA/E	PLA/AGF	PLA/HNF
Bending	Modulus (GPa)	3.36 ± 0.02	10.15 ± 0.25	9.28 ± 0.09	5.78 ± 0.09
	Strength (MPa)	102 ± 1	171 ± 5	138 ± 4	102 ± 2
	Elongation (%)	3.5 ± 0.0	1.8 ± 0.1	1.6 ± 0.1	2.4 ± 0.2
Tensile	Modulus (GPa)	3.61 ± 0.07	10.48 ± 0.28	9.81 ± 0.08	5.89 ± 0.12
	Strength (MPa)	72 ± 1	122 ± 2	116 ± 6	73 ± 1
	Elongation (%)	7.5 ± 1.2	3.3 ± 0.1	3.4 ± 0.3	3.0 ± 0.1
Impact	Resilience (kJ/m^2)	30 ± 5	29 ± 2	32 ± 2	14 ± 2

Table 7. Mechanical properties of PLA and PLA composites reinforced by a conventional E-glass fibres (PLA/E), alterable glass fibres (PLA/AGF) and hemp natural fibres (PLA/HNF) (30 wt%)

The mechanical data are summarized on Table 7. The results show that the presence of hemp natural fibres within the PLA slightly increased the tensile and bending moduli but did not improve other mechanical properties (strength and elongation). Moreover a lower resilience was obtained for the HNF reinforced biocomposites compared to unreinforced

PLA. The presence of glass-based fibres led to a significant increase in bending and tensile moduli and strengths and a decrease in corresponding elongations. The impact properties were not influenced by the presence of glass-based fibres. However, AGF-fibres result in lower mechanical properties than E-fibres.

4.1.2.3 Ageing resistance of biocomposites

One of the scientific problems is to be able to maintain consistent properties throughout the in-service use of the biocomposites, while also being able to trigger their ultimate biodegradation/composting at the end of their life, as shown in Figure 8.

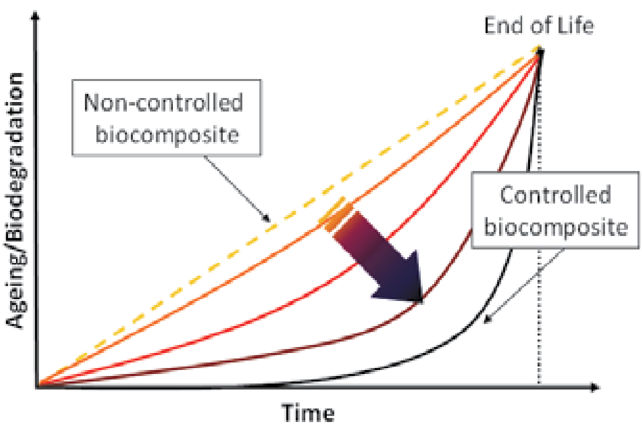


Fig. 8. Control of ageing and biodegradation kinetics of materials

It is well known that the main degradation mechanism of PLA is hydrolysis, which increases markedly above Tg and with ageing time due to the formation of hydrophilic groups such as alcohols and acid functions (Li & McCarthy, 1997). In addition, crystallinity and the presence of microvoids or porosity may affect water permeation (Drumright et al, 2000). Several ageing tests have been developed to analyze the evolution of material properties, among them accelerated ageing tests. For the present study biocomposites were conditioned within an autoclave with 100 RH% and at 65°C (above Tg), simulating several years of in-service use. Figure 9 shows the evolution of mechanical properties together with water absorption and crystallinity rate against ageing time. A significant decrease can be observed for all mechanical properties, with the lowest decrease being obtained for PLA/E biocomposites and the highest for PLA/AGF ones. With regard to water absorption, the presence of E-glass fibres may decrease the water content (0.61% compared to 1.37% for PLA) whereas the alterable fibres did not change the water content. A huge absorption rate was observed for HNF reinforced biocomposites (4.35%) due to the intrinsic hydrophilic character of these fibres. Whatever the biocomposite, an increase in crystallinity rate can be observed at short ageing time (+20% after an ageing time of 24h), then a plateau is obtained except for HNF reinforced PLA.

4.1.2.4 Biodegradation of biocomposites

Different degradation tests were performed on the PLA and biocomposites.
One of them was the previously described BOD test (see § 3.1.2.2.4), which showed no influence of the presence of alterable glass fibres (AGF) either on the latency time (about 10 days) or on the final degradation rate after 28 days (about 15 %) of PLA.

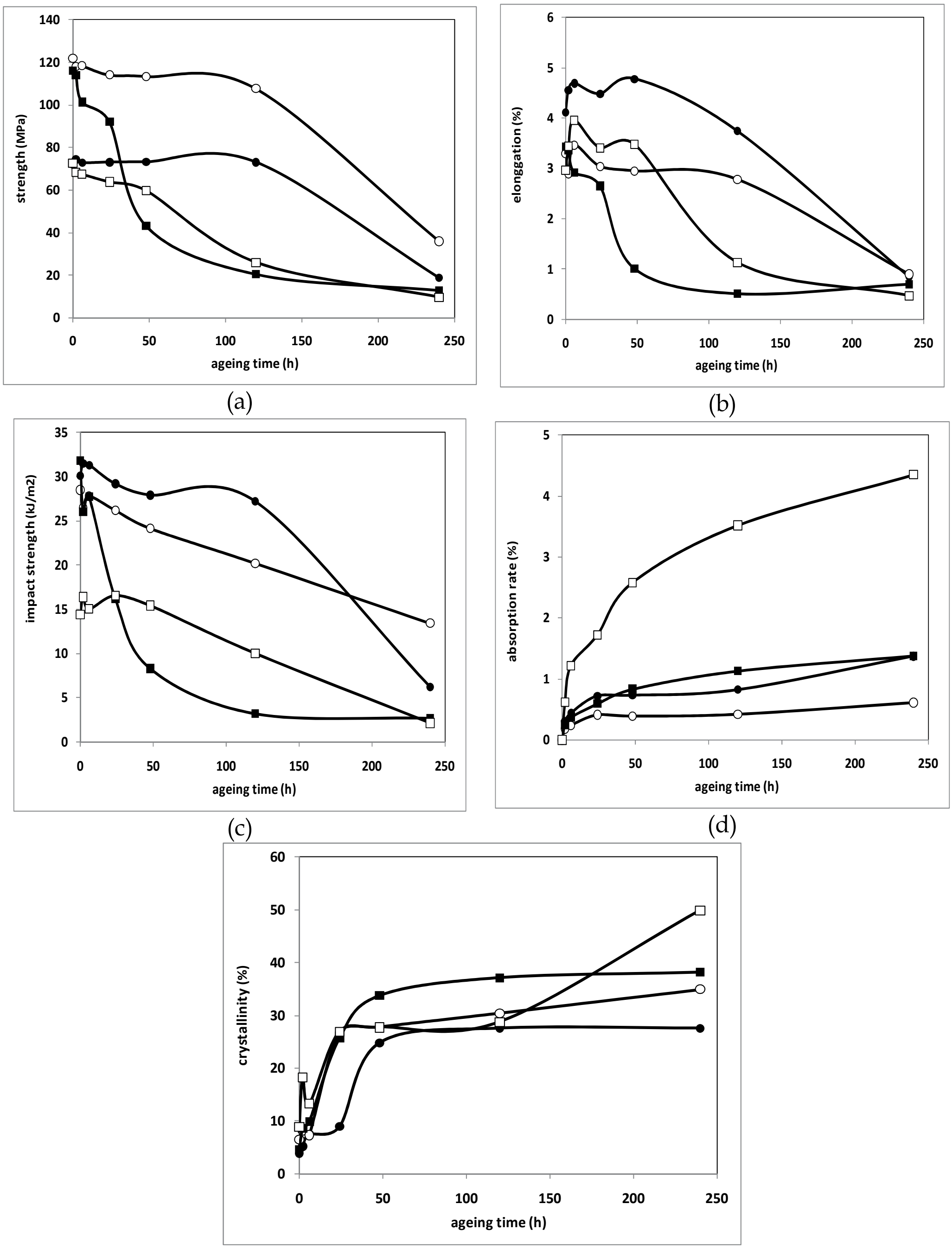


Fig. 9. Variation of (a) ultimate strain, (b) ultimate stress, (c) impact strength, (d) water absorption rate and (e) crystallinity rate of (●) PLA and PLA composites reinforced by by (O) conventional E-glass fibres (PLA/E), (■) alterable glass fibres (PLA/AGF) and (□) hemp natural fibres (PLA/HNF) (30 wt%) with ageing time (65°C; 100 %RH)

Another test was performed according to XPU 44-163 standards and gave the mineralisation rate of PLA and biocomposites (Figure 10). It was observed that mineralisation rate of PLA does not stop increasing with a value of 170 mg CO_2/g of PLA after 49 days with a final pH of the soil of 4.24. In the case of the biocomposites, a decrease in this mineralisation rate can be observed with various behaviours according to the fibre nature. In presence of AGF a plateau is reached after 25 days (120 mg CO_2/g of composite) and acidification of the soil is observed (final pH at 4.06) which may induce a decrease in microbial activity. The lowest mineralisation rate, together with acidification of the soil (final pH at 3.91,) was obtained in the presence of HNF (60 mg CO_2/g of composite).

One purpose of this study was to propose glass formulation that may buffer the soil to avoid acidification and therefore the decrease in microbial activity.

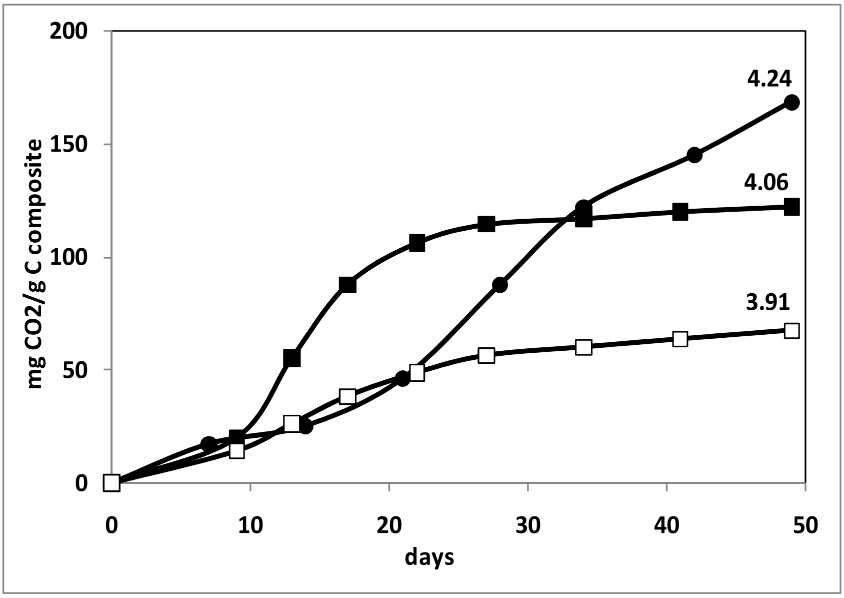


Fig. 10. Mineralisation rate and final pH of the soil for (●) PLA and PLA composites reinforced by (■) alterable glass fibres (PLA/AGF) and (□) hemp natural fibres (PLA/HNF) (30 wt%)

4.1.2.5 Further studies

Current studies are focussing on improving both the glass formulation and the PLA-based matrix properties. Concerning the glass, new glass compositions are being developed to control either the mechanical properties of the alterable fibre in order to be comparable to conventional E-glass fibres (with a Young's modulus about 73 GPa) or the alteration mechanism. For the PLA-based matrix, investigations are being carried out to increase impact properties as well as durability by incorporating impact modifiers and/or blending with other biodegradable polymers that are more ductile and less hydrophilic than PLA.

4.1.3 Polylactic acid-based films for textile applications

This second case study concerns PLA-based films for some textile applications, especially disposable safety workwear. The required performances (among them tear resistance, weldability, perforation, thermal resistance, barrier properties) were achieved by incorporating nanoclays and specific additives.

Various mechanical behaviours were obtained according to the nature of the nanoparticles and their surface modifications. A fibrillar sepiolite and a lamellar montmorillonite (called MMT) were compared for a content of 2.5 wt%. They were respectively treated with a silane coupling agent (γ-methacryloxy-propyltriethoxysilane) and a quaternary ammonium salt. PLA (PLA4032D© provided by Nature Works LCC, USA) was plasticized (18 wt% of tributylacetylacetate from Sigma Aldrich). A chain extender (styrene-co-glycidyl methacrylate, trademarked Joncryl 4368© from BASF) was also added. A three-step process was involved including (i) twin screw extrusion to obtain a PLA/clay masterbatch (85/15 w/w), (ii) dilution and addition of plasticizer and chain extender through twin screw extrusion and (iii) blowing extrusion to obtain 50µm thick films.

The first analysis of the behaviour was performed using the tensile tests performed for each formulation. These tests were carried out using an optical extensometer combining a Charge Couple Device (CCD) camera associated with Digital Image Correlation (DIC) software. This revealed the evolution of the in-plane strains (transverse and longitudinal components) (Figure 11). Table 8 summarizes all mechanical characteristics. While it can be observed that a lower yielding stress was obtained for PLA films filled with sepiolite clay, on the other hand they showed higher work hardening (K).

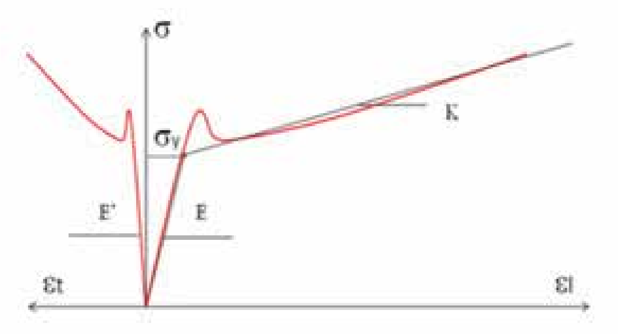


Fig. 11. Evolution of the in-plane strains (transverse and longitudinal components)

	E (MPa)	σy (MPa)	K
PLA	1100 (10)	26 (11)	1 (8)
PLA/sepiolite	390 (10)	16 (8)	17 (3)
PLA/mod-sepiolite	600 (14)	15 (7)	17 (3)
PLA/MMT	700 (10)	21 (4)	12 (5)
PLA/mod-MMT	1100 (10)	27 (2)	13 (5)

Table 8. Longitudinal mechanical characteristics obtained from loading curves of stress vs deformation (100 mm/min, sample 150x50x100 mm^3) for PLA and PLA composites reinforced by unmodified and modified (mod) sepiolite and montmorillonite – (): variance

4.1.4 Polylactic acid-based foam products

Obtaining PLA-based foam products is of major industrial interest, in order to replace high-mass products by lighter bio-based ones with identical performances. They can be considered as interesting alternative candidates.

The objective of the studies concerned was to optimize either the processing conditions (extrusion flow rate, temperature, cooling system) or the material formulation (content of chemical blowing agent, PLA characteristics) for maximum foam expansion and good mechanical performances.
Results show that the nature of the PLA (PLA 7000D© and PLA4032D© provided by Nature Works LCC, USA) had no major effect on the void content of the foams extruded under the same conditions (screw range temperature: 150-180°C) (Figure 12). The density varied between 893 and 879 kg/m³ for a CBA content of 2 wt% (corresponding to a density reduction of 29 % and 30 %, respectively). Moreover, the void content increased gradually (the foam density decreased) with the CBA content, regardless of the PLA type or temperature profile. This evolution is related to the amount of gas formed and available for the expansion process (Klempner & Sendijarevic, 1991). Similar results have been reported for extruded foams based on polyolefins (Lee, C.H. et al, 2000; Lee, S.T., 2004, 2008).

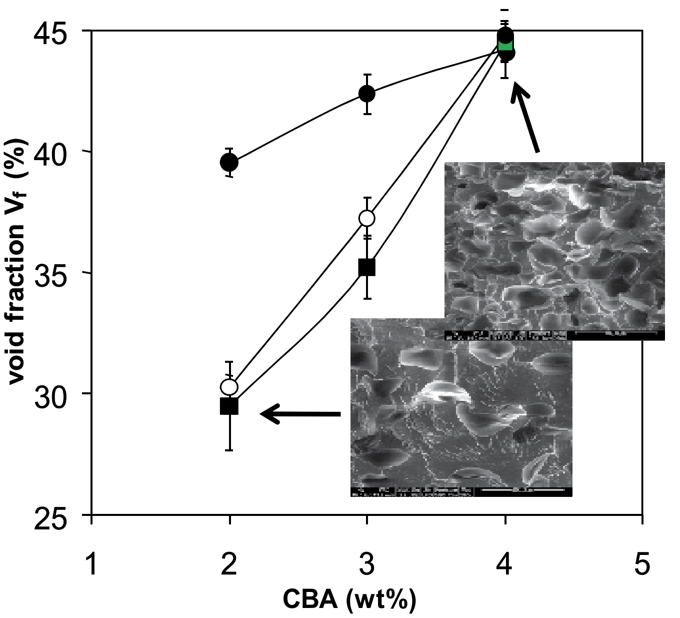


Fig. 12. Void fraction of PLA foams as a function of the chemical blowing agent content (CBA), the processing conditions (screw range temperature : ● 130-190°C; ■,O 150-180°C) and the PLA nature (O PLA4032D©, ●,■ PLA7000D©) and corresponding microstructures (ESEM observations; magnitude 250X)

In addition, both PLAs investigated provided foams with homogeneous cellular structures (polydispersity PDI close to 1) (Table 9). In all cases, the open-cell ratio was low, below 26 %. Similar results have been reported for polyolefin foams (Klempner & Sendijarevic, 1991; Ray & Okamoto, 2003; Ema et al, 2006). An increase in the open-cell ratio with the CBA content is observed which is related to the amount of gas release. This trend has already been reported by other authors (Klempner & Sendijarevic, 1991; Lee, C.H. et al, 2000). A higher open-cell ratio range is obtained for PLA4032D© (ratio between 19 and 27% for CBA content between 2 and 4 wt%) compared to PLA7000D© (ratio between 11 and 19%). This is related to the higher temperatures involved in the temperature profile used for PLA4032D© inducing a higher gas yielding as well as a lower PLA viscosity The cell density of the PLA7000D©-based foams processed with a screw temperature range 130-190°C was significantly higher than that of PLA4032D©-based foams processed with the screw temperature range 150-180°C. This is undoubtedly related to gas loss through the first barrel zones during extrusion-foaming of the PLA4032D©. Moreover, for PLA7000D©, the cell density decreases with the CBA content as the cell size increases with no variation in the cell wall thickness. For PLA4032©, a slight increase in cell density is observed with a non monotonous variation in cell size and a significant decrease in the cell wall thickness. It can be assumed that several competitive mechanisms may occur: (i) the increase in gas yielding and decrease in viscosity due to barrel temperatures, (ii) the plasticization induced by the gas products during decomposition and (iii) the presence of a higher content of nucleating agents present in the CBA masterbatch. Complementary work is in progress to evaluate the relative contributions of these mechanisms. The average cell diameter and cell-wall thickness in the present study were similar to those reported for polyolefin foamed with CBA (Klempner & Sendijarevic, 1991; Lee, C.H. et al, 2000), but significantly higher than those reported by other authors for microcellular PLA foams (Ray & Okamoto, 2003). Nevertheless, the cases reported in the literature concern mainly physical foaming processes, and PLA modified by nanofillers (which may act as cell nucleating agents) and/or chain extenders.

Finally, for both types of PLA investigated, the increase in CBA content led to a reduction in stresses at yield and break in tension (Table 10), due to the increase in void content (reduction of the effective sample cross-section). On the contrary, elongation at yield and break were low and independent of the CBA content. Similar results were reported for PVC and PUR-based foams (Kabir et al, 2006; Lin, 1997).

PLA	CBA1 (%wt)	d_n (µm)	d_w (µm)	PDI	N_c (cells.cm^{-3}) ×10^5	δ (µm)	C_o (%)
7000 D©	2	90 (2)	105 (1)	0.86 (0.01)	11.25 (0.78)	48 (1)	10.91 (0.24)
	3	95 (2)	104 (1)	0.91 (0.01)	10.13 (0.70)	46 (1)	14.72 (1.08)
	4	107 (2)	122 (2)	0.88 (0.01)	7.38 (0.51)	49 (1)	19.22 (0.69)
4032 D©	2	134 (3)	144 (2)	0.93 (0.01)	2.72 (0.19)	95 (2)	19.12 (1.47)
	3	125 (3)	174 (2)	0.72 (0.01)	4.02 (0.28)	71 (2)	24.49 (1.3)
	4	130 (3)	152 (2)	0.86 (0.01)	4.19 (0.29)	58 (1)	26.76 (1.11)

Table 9. Cell dimensions (d_n, d_w), cell density (N_c), cell size polydispersity (PDI), cell-wall thickness (δ) and open-cell ratio (C_o) as function of CBA content – PLA 7000D© screw temperature range 130-190°C and PLA 4032D© screw temperature range 150-180°C; screw speed 30 tr/min; die temperature 195 °C; free cooling - ( ) : standard deviation

4.2 Polyhydroxyalkanoates

Like polylactic acids, polyhydroxyalkanoates (PHAs) are aliphatic polyesters (Figure 13) produced via fermentation of renewable feedstock. Whereas PLA production is a two-stage process (fermentation to monomer followed by a conventional polymerization step), PHAs are produced directly via fermentation of carbon substrate within the microorganism. The PHA accumulates as granules within the cytoplasm of cells and serves as a microbial energy reserve material. PHAs have a semicrystalline structure, the degree of crystallinity ranging from about 40% to around 80% (Averous, 2004).

PLA	CBA (wt%)	Void fraction (%)	σ_{max} (MPa)	ε_{max} (%)	σ_r (MPa)	ε_r (%)
7000D©	2	42 (1)	36.00 (2.84)	8.41 (0.32)	33.38 (2.18)	9.95 (0.71)
	3	45 (1)	25.65 (1.33)	8.84 (0.39)	21.51 (1.72)	11.59 (0.92)
	4	47 (1)	23.71 (0.90)	8.37 (0.55)	20.49 (1.08)	10.28 (1.16)
4032D©	2	34 (1)	40.54 (3.27)	10.24 (0.89)	37.91 (3.90)	11.51 (1.23)
	3	41 (1)	30.68 (2.40)	8.61 (0.70)	27.14 (3.44)	10.43 (1.11)
	4	48 (1)	26.77 (1.24)	9.44 (1.36)	22.87 (1.12)	11.50 (1.60)

Table 10. PLA-foams tensile properties. Processing conditions: PLA 7000D© screw temperature range 130-190°C and PLA 4032D© screw temperature range 150-180°C; screw speed 30 tr/min; die temperature 195 °C; free cooling. (σ_{max} : yield stress, ε_{max} : elongation at yield stress, σ_{r} : stress at break, ε_r : elongation at break) - ( ) : standard deviation

Table 11 shows the generic formula for PHA where x is 1 (for all commercially – relevant polymers) and R can be either hydrogen or hydrocarbon chains of up to around C16 in length. A wide range of PHA homopolymers, copolymers and terpolymers have been produced, in most cases at the laboratory scale. A few of them have attracted industrial interest and been commercialized in the past decade.

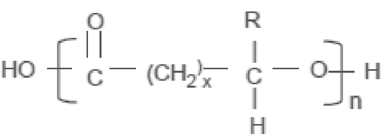


Fig. 13. PHA molecule

PHA	full name	x	R
PHB	Poly(3-hydroxybutyrate) P(3HB)	1	$-CH_3$
PHV	Poly(3-hydroxyvalerate) P(3HV)	1	$-CH_2CH_3$
PHBV	Poly(3-hydroxy butyrate-co-valerate) P(3HB-co-3HV)	1	$-CH_3$ and $-CH_2CH_3$
PHBHx	Poly(3-hydroxy butyrate-co-hexanoate) P(3HB-co-3HHx)	1	$-CH_3$ and $-CH_2CH_2CH_3$
PHBO	Poly(3-hydroxy butyrate-co-octanoate) P(3HB-co-3HO)	1	$-CH_3$ and $-(CH_2)_4CH_3$

Table 11. Examples of structures of PHA

Like PLA, PHA is sensitive to processing conditions. Under extrusion, a rapid decrease in viscosity and molecular weight can be observed due to macromolecular linkage by increasing the shear level, the temperature and/or the residual time (Ramkumar & Bhattacharia, 1998). The kinetics of enzymatic degradation varies according the crystallinity and processing history (Parikh et al, 1998).

At present, packaging (bags, boxes), agriculture mulching films and personal care items (razors, tooth brush handles) are the most important market for PHA. In the future, the applications will become broader: building, textile, transportation, electronics, houseware, etc.

4.3 Biopolyesters obtained from petrochemical products

Several biopolyesters can be obtained from petrochemical products, among the most commonly used is polycaprolactone (ring opening polymerization of caprolactone resulting from moderate oxidation of cyclohexanone). The other biopolyesters are produced through condensation reactions between diols and diacides, for example polybutylene succinate (PBS), polybutylene adipate terephthalate (PBAT). All these biodegradable polymers have interesting ductile properties, and are thus frequently combined with rigid PLA.

PCL is widely used as a PVC solid plasticizer or for polyurethane applications. There are also some applications based on its biodegradable character in the controlled release of drugs and soft compostable packaging.

5. Conclusion

The progress made in the field of environmental-friendly biodegradable polymers and composites over the past ten years has been impressive. A large number of companies are now involved in this area, producing a wide range of products. There are also major ongoing advances in research and development, contributing to the increased attractiveness of chemical sciences and technology for a new generation of scientists and engineers. All in all, these developments have converted bio-based polymers and composites from a minor niche into a mainstream activity. However, the challenges that need to be successfully addressed in the years and decades to come are the lower material performance of some bio-based polymers, the control of the lifetime during in-service life regardless of their end-of–life biodegradation, their relatively high production and processing costs, and the need to minimize the use of agricultural land and forests, thereby also avoiding competition with food production and adverse effects on biodiversity and other environmental impacts.

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CEA (Commissariat à l'Energie Atomique), Marcoule; Ecole des Mines de Douai, Douai) involved in the various studies presented in this chapter.

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Geochemical Risk Assessment Process for Rio Tinto's Pilbara Iron Ore Mines

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1. Introduction

Acid and Metalliferous Drainage (AMD) is a major environmental risk that should be regularly assessed at all new and existing iron ore mine sites. AMD can often be reduced or prevented by appropriate mine plans but where not managed properly, can lead to costly collection and treatment programs that must function for many decades. This is particularly evident at many historical and abandoned mine sites where the AMD was not identified prior to mining.

Whilst the release of acidity alone can have major impacts, the dissolution of metals (such as iron, aluminium, manganese, cadmium, copper, lead, zinc, arsenic and mercury) from surrounding country rock can also have significant downstream impacts on the environment. The water quality may impact on human health and thus increase public and regulatory focus and concern. Ultimately the 'social licence to operate' may be at risk.

Metalliferous drainage typically requires, at a minimum, low-pH conditions on a microscopic scale as a mechanism to initially solubilise contaminants. If the sulfide-bearing rock also has sufficient neutralising capacity, the acid generated is subsequently neutralised. However despite neutralisation, concentrations of some contaminants do not precipitate at near-neutral pH (ie. zinc, arsenic, nickel, and cadmium). Instead these contaminants remain in solution resulting in low-quality drainage. In cases where there has been sufficient neutralisation to remove all metals the water can still have elevated concentrations of sulfate resulting in elevated salinity. It is therefore important to adequately geochemically assess all material at a mine site to ensure that all aspects of AMD risk are considered.

A crucial step in leading practice management of AMD is to assess the environmental, human health, commercial and reputation risks as early as possible (CoA 2007). Reactive mineral waste can cause harm by: degrading water quality causing human health or ecological impacts; inhibiting vegetation establishment, posing a direct exposure risk to animals and humans; and degrading air quality through dust or gas emissions.

Rio Tinto and its subsidiary Rio Tinto Iron Ore (RTIO) have developed standards, strategies, procedures, management plans and guidance notes that can be used to assess AMD risk for mine sites. This paper summarises how these documents have been integrated and how site specific information is assessed for AMD risk at Rio Tinto's Iron Ore (RTIO) mines in the Pilbara region of Western Australia. Guidance for conducting ecological risk assessments

(Linkov et. al. 2002) or water quality risk assessments (ANZECC 2000) are more prevalent than that for AMD and geochemical risk assessments. RTIO's four stage process to evaluate the AMD and geochemical risks for a mine site are unique and comprehensive. This process could be used as a guide to conduct AMD and geochemical risk assessments at other mining operations.

2. RTIOs mining operations in the pilbara region

2.1 Location

Within the Pilbara of Western Australia, RTIO manages mines, ports and rail infrastructure for Hamersley Iron Pty Ltd (Greater Tom Price, Greater Paraburdoo, Marandoo, Greater Brockman and Yandicoogina (Yandi)), Robe River Iron associates (West Angelas and Mesa J (Pannawonica) and Hamersley HMS Pty Ltd (Hope Downs 1). Hereafter RTIO refers to all these groups.

Iron ore is mined in open cut truck and shovel operations using drilling and blasting. Blast holes are drilled by rotary and hammer drill rigs on 10 or 15 m benches designed to suit the geology or equipment of the individual mine. Iron ore from inland mine sites is transported via the 1,481 km railway network to port facilities located at Dampier and Cape Lambert (Fig. 1).

2.2 Mineral waste risks

Mineral waste is composed of bedrock or unconsolidated sediments that are disturbed or exposed by mining. Mineral waste can also be composed of mineral residue generated by the processing of ore. The environmental exposure hazards of reactive mineral waste whose innate physical, chemical or biological properties could now or in the future pose harm, are a risk that RTIO endeavour to manage, using best practice management techniques. RTIO also invests significantly in research and development in this area. During the 2009 financial year RTIO directly invested $1.2 million (Aus) into mineral waste research for the Pilbara. This research has included modelling of final pit void water quality, bioremediation, cover research, waste dump designs and geochemical characterisation (Green 2009).

Whilst not a risk at every mine site in the Pilbara, it is particularly important to evaluate the risk for Acid Rock Drainage (ARD), contaminants soluble at neutral pH, salinity and organic compounds (including spontaneous combustion hazards). Although not necessary a geochemical risk, fibrous minerals are also an important consideration for mining operations in the Pilbara.

2.3 Geological setting

Banded Iron Formation (BIF) derived iron deposits occur where BIF has been locally enriched in situ. BIF-derived iron deposits may be hosted in the Marra Mamba Iron Formation, or in the Joffre and Dales Gorge members of the Brockman Iron Formation (Fig. 2). Of the BIF-derived iron deposits, only those associated with the Dales Gorge member of the Brockman Iron Formation are likely to occur in close proximity to the potentially carbonaceous and sulfide-bearing Mount McRae Shale (MCS). Less reactive black shale can also be found in thinner bands than the MCS in the Footwall zone, Dales Gorge, Jeerinah Formation, Wittenoom Formation, Nanutarra Formation, Ashburton Formation and Whaleback Shale members.

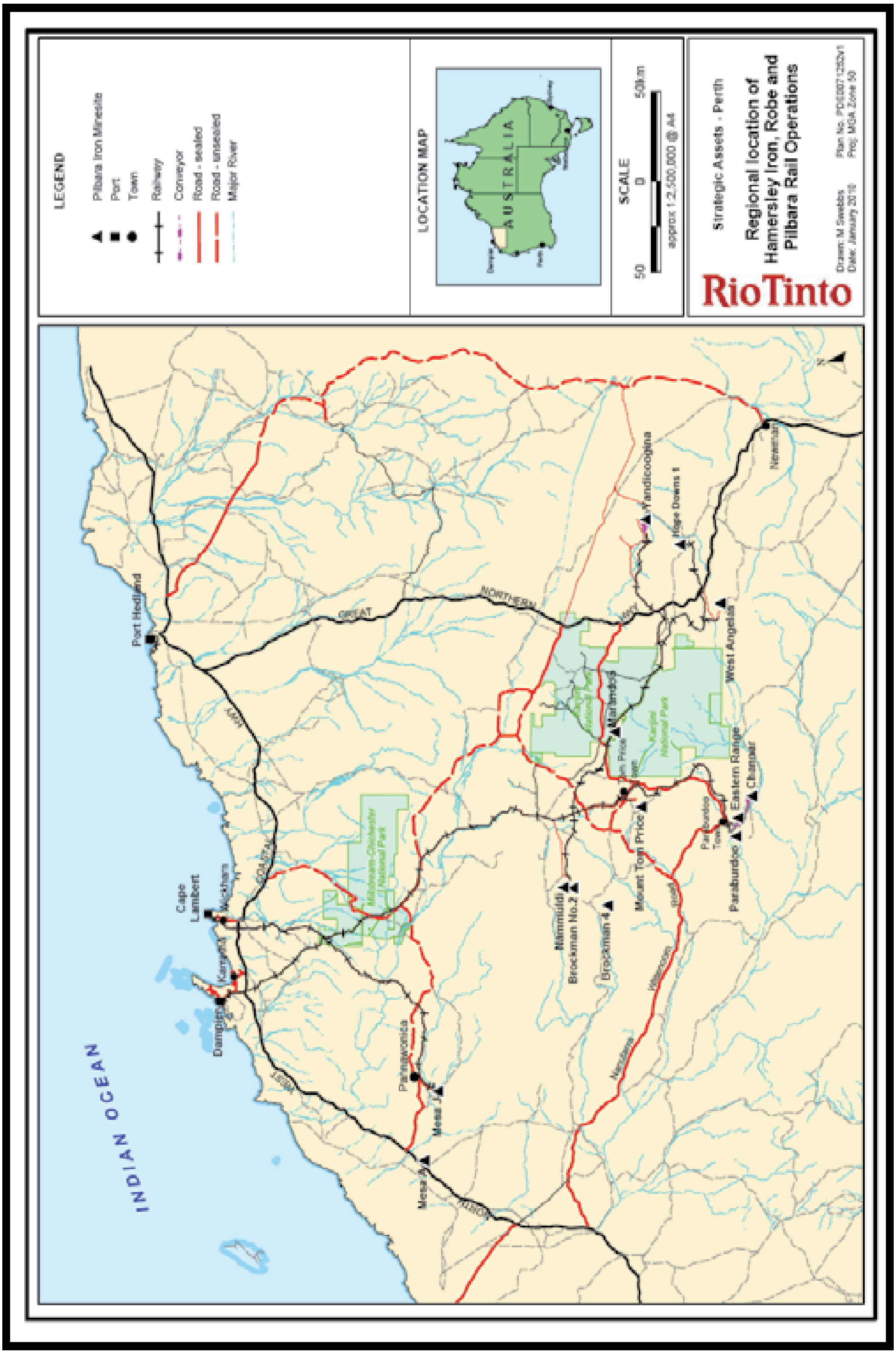

Fig. 1. The location of RTIO's Pilbara operations.

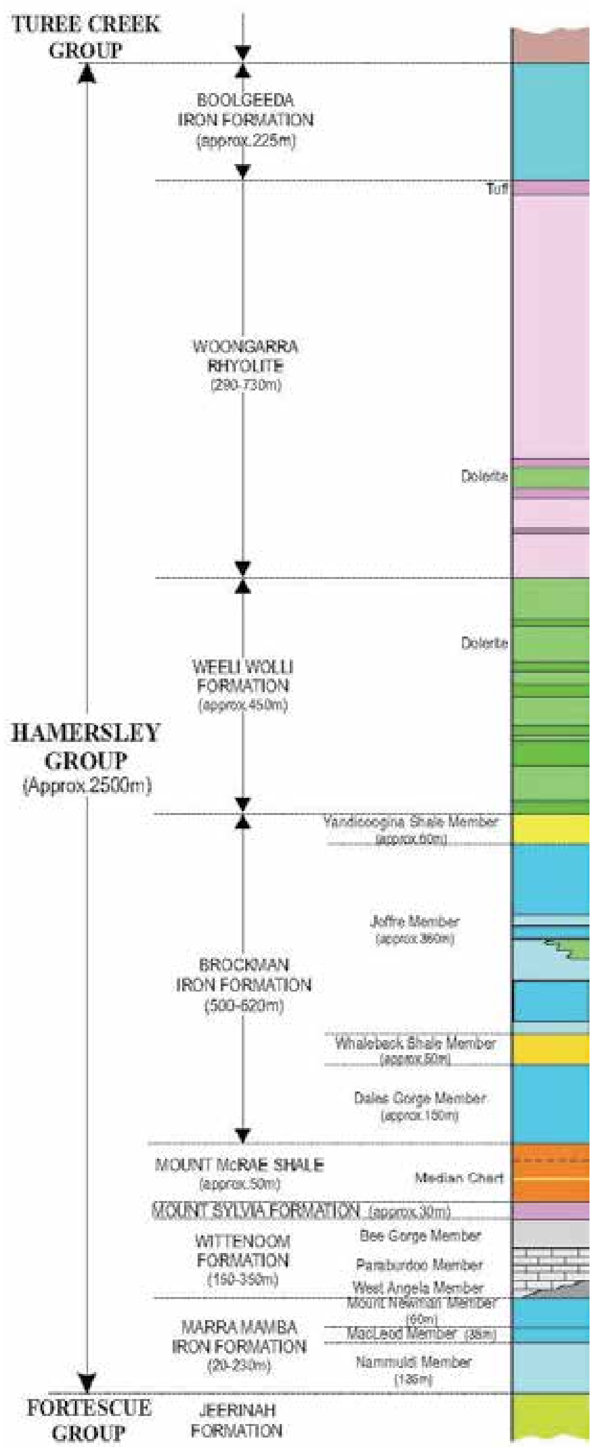

Fig. 2. Stratigraphic column of the Hamersley Group.

Enriched Marra Mamba Formation ore is most commonly found in the Newman member and whilst carbonaceous black shale is not typically associated with these units, pyrite can be found in all three members of the Marra Mamba Formation.

Detrital Iron Deposits (DIDs) and Channel Iron Deposits (CIDs) occur where enriched BIF has been exposed at the ground surface and material has been eroded and/or transported and redeposited. Detrital iron ore units, including unconsolidated scree, hematite conglomerate and CIDs of pisolite also occur in alluvial valleys. Sulfidic material can be associated with carbonaceous lignites and siderite interbedded with the iron ore deposits.

3. Corporate guidance

A number of Rio Tinto and RTIO documents are relevant for the management of AMD at mine sites (Fig. 3). There are 10 Environmental standards that are regularly audited against for compliance. Non compliance is tracked as audit actions within Rio Tinto. The ARD Standard applies to the full mine life cycle from exploration through to post-closure. It covers planning, implementation and operation, and performance monitoring. Rio Tinto have also embarked on an extensive risk review process that involved internal and external geochemical and hydrogeological experts visiting every Rio Tinto mine and project site with a significant potential AMD risk. Sites are initially screened using the Rio Tinto Hazard Screening Protocol to identify those mine sites with a significant potential AMD risk (Richards *et. al.* 2006). The risk reviews provide commentary on how each site is managing the hazard, identifies areas that need further investigation, and identifies management improvements needed to reduce the overall risk. Action plans are developed and tracked within Rio Tinto based on the findings from the risk reviews (An AMD risk review was completed in 2005 at the Pilbara operations).

In response to the Rio Tinto Standards, RTIO developed a mineral waste strategy and subsequently developed the Mineral Waste Management Plan (MWMP) that is applicable for every Pilbara mine site. This plan has two major sections. The first section describes the actions to be taken before mining commences at resource drilling, order of magnitude studies, pre-feasibility studies, feasibility studies to mine development. The second section of the plan describes the actions to be taken during mine operations and includes planning, operational and monitoring considerations. Extensive guidance is provided within the appendix of the plan including:

- Hyperlinks to all previous mineral waste related reports;
- Detailed description of known geological risk;
- Instructions for the inclusion of mineral waste information in the Resource block models;
- Analysis of mineral waste geochemical risk;
- Analysis of unconsolidated sediment geochemical risk; and
- Site water quality compliance criteria.

This plan is relevant for all RTIO mines in the Pilbara and is used to regularly monitor and assess AMD risk. The requirement for a risk assessment is an action within the plan.

If the risk assessment and work undertaken to comply with the MWMP identifies a significant AMD risk then the Spontaneous Combustion and ARD (SCARD) management plan needs to be implemented at the mine site. This plan describes the actions that need to be taken by long term planning, site planning, geology, survey, operational planning, blasting, hauling, hydrogeology, environment, health and safety and the mineral waste management team to reduce AMD risk at the mine site. Regular meetings are held at the mine with a representative from each group to discuss compliance with the plan and AMD risk. Extensive guidance is provided within the appendix of this document and includes:

- Hyperlinks to all previous mineral waste related reports;
- Detailed description of known geological risk;
- Dump designs;
- Rehabilitation and closure; and
- Contingency planning

Links are made in the SCARD management plan to the site specific documents for each mine site. These documents are mostly safe work procedures and health guidance notes that are specific to individual operations.

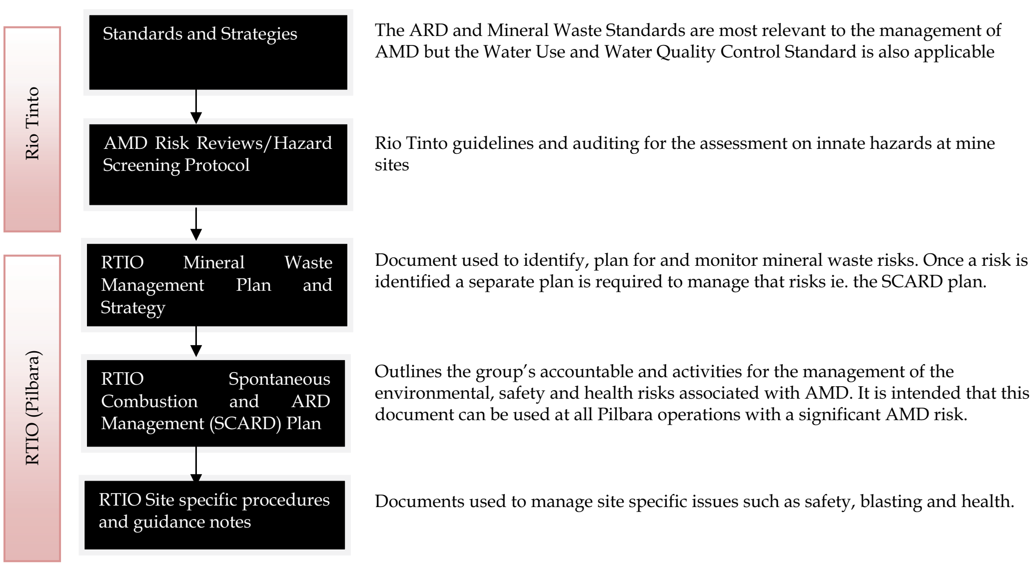

Fig. 3. Significant corporate guidance documents from Rio Tinto and RTIO for the management of AMD.

4. Risk assessment process

A thorough AMD and geochemical risk assessment enables the study team to analyse the issues, prioritise and make informed decisions. Continual awareness of risk management enhances and encourages the identification of greater opportunities for continuous improvement through innovation (AS/NZS ISO 31000:2009). Risk assessment also assists decision-makers to deal with uncertainty. The risk assessment process is designed to minimise uncertainty associated with potential and actual risks and hazards.

The objectives of the RTIO AMD and geochemical risk assessment process are specifically:

- To identify the hazards and resultant risks to the environment from the project as a whole;
- To identify opportunities to manage or avoid AMD upfront;

- To provide a rigorous basis for decision making and planning;
- To evaluate and prioritise the risks and identify management measures to mitigate the risks;
- To reduce unexpected occurrences;
- To reduce business risk and operational expense;
- To enhance due diligence studies, governance, stakeholder relationships and business reputation;
- To improve the health and safety of employees and the public;
- To identify research and development opportunities; and
- To minimise long term post closure risks, liabilities and environmental impacts.

There are four stages to the RTIO AMD and geochemical risk assessment for a deposit. The first stage can be completed by anyone within the business possessing good knowledge of the deposit. However the next three stages of the risk assessment require specialist AMD expertise. Progressively more knowledge is required through each of the stages to analyse the risk. Rio Tinto and RTIO retain much of this expertise internally.

4.1 Stage 1: preliminary AMD hazard score

During the order of magnitude or exploration phase of a mining project a preliminary assessment of AMD risk can be made based on the guidelines provided by Rio Tinto. This Hazard Screening Protocol ranks the hazard at a site based on the innate physical and chemical setting and no commentary is provided on the sites management measures. Readily available data is used to assess the likelihood for a significant AMD source at a site, as well as determining if there are dispersal pathways that could create significant down gradient environmental impacts. Numerical values are assigned for each of the following categories encountered at each site, according to a rating of relative influence:

- Geology (45%)
- Ore deposit type (30%)
  - Host and country rock neutralisation potential (10%)
  - Known ARD issues on site (5%)
- Incipient ARD Risk (5%)
  - Operational age (5%)
- Scale of Disturbance (25%)
  - Total waste stored on site (15%)
  - Footprint of disturbed area (10%)
- Transportation pathways (10%)
  - Water availability (7%)
  - Metal release to the environment (3%)
- Sensitivity of the receiving environment (15%)
  - Proximity to perennial/ephemeral water bodies (5%)
  - Alkalinity of water body or groundwater (5%)
  - Distance to closest protected/permanently inhabited area (5%)

A group of 10 experts were involved with the development of these factors and their weightings. The weighting factors were further refined by ranking a series of well known mines and ensuring it corresponded with the professional judgment of those experts involved.

Some modifications have been made to the broad Rio Tinto Hazard Score to make it more applicable for the Pilbara. These changes and a general description of the major factors are in the following sections. An example of a preliminary AMD hazard score assessment for a site is demonstrated in Fig. 4.

Project Name	Example site	Version Date: 5/03/10
Assessment Date	12/11/2010	Version Number: 2
Compiled by	Ros Green	
Final ARD Hazard Assessment	MODERATE	

RTIO AMD Hazard Score

1. Preliminary Assessment (Order of Magnitude/Exploration)

A. Preliminary Geology Hazard

	Select Relevant Option Below	Score	Option Details
Ore Deposit Type	C) Enriched Marra Mamba Formation and Joffre Member, and/or channel and detrital ore bodies mined below the water table (un-oxidised lignite and black shales other than Mt McRae may be present). Enriched Dales Gorge Member mined above the water table only	14	No PAF material expected
Host & Country Rock Neutralising Potential	None (<5%)	10	Minor calcrete in project area
Brownfield's / Greenfields	Brownfield		
Known AMD Issues on Site	No	0	
		Geology Hazard Score	24

Complete following sections

B. Incipient AMD Risk

	Select Relevant Option Below	Score
Operation Age	< 5 years	5

**By default, all new projects should receive a <5 years value*

C. Scale of Disturbance

	Select Relevant Option Below	Score
Total Waste Stored	50 - 250 million tonnes	5
Footprint	250 - 1000 hectares	6

D. Transport Pathways

	Select Relevant Option Below	Score
Project / Exploration?	No	
Precipitation / Areal Potential Evapo-transpiration Ratio	1/10 to 1/3 ratio_mining below the water table in a rock mass that is connected to a regionally significant aquifer	3

**All new projects should respond Yes to Project / Exploration*

E. Sensitivity of Receiving Environment

	Select Relevant Option Below	Score
Distance to Perennial Water Bodies	>2000 metres	0
Distance to Ephemeral Water Bodies	>2000 metres	0
Alkalinity	>35 mg/L	1
Distance to closest protected / permanently inhabited area	<500 metres	5

Preliminary Hazard Assessment

Preliminary Hazard Score	49
Preliminary Risk Assessment	MODERATE

Fig. 4. Example of the use of preliminary AMD Hazard score to assess a site.

4.1.1 Geology

Most RTIO deposits in the Pilbara are ore bodies that exist under reducing conditions but whose genesis is not directly related to sulfide mineralisation. Supergene enriched BIF or

Detritals that are above the water table and have been exposed to long term weathering are unlikely to contain sulfides within the ore however sulfide bearing shale or lignite may be inter-bedded with or lie stratigraphically below the ore body. The risk ranking score for the RTIO deposits in the Pilbara assigns a higher risk to below water table mining (Table 1). Mining of ore within the Dales Gorge Formation is assigned a higher score than other ore body stratigraphies due to the underlying and typically reactive black MCS.

Ore Deposit Type	**Score**
A) Formation by active surficial processes in equilibrium with the atmosphere.	0
B) Enriched Marra Mamba Formation or Joffre Member, and/or channel and detrital ore bodies mined above water table only (no Mt McRae Shale present and all rock types likely oxidised).	7
C) Enriched Marra Mamba Formation or Joffre Member, and/or channel and detrital ore bodies mined below the water table (un-oxidised lignite and black shales other than Mt McRae may be present). Enriched Dales Gorge Member mined above the water table only.	14
D) Enriched Dales Gorge Member mined below the water table (un-oxidised Mt McRae shale likely present)	19
E) Formation is directly associated with low-grade (< roughly 10 % total sulphur) acid generating sulphide mineralisation (not applicable to Pilbara Iron deposits).	23
F) Formation is directly related to high-grade (> roughly 10% total sulphur) or very reactive acid generating sulphide mineralisation (not applicable to Pilbara Iron deposits).	30

Table 1. Hazard scores based on the geology of the deposit.

Enriched and un-enriched BIF mined in the Pilbara typically has a low neutralising potential. Shales also typically have low neutralising potential. However calcretes mined in Detrital deposits can have readily available neutralising potential (Acid Neutralising Capacity or ANC of 265-660 kg H_2SO_4/t). In addition dolomite within the Wittenoom Formation (ANC of 301-885 kg H_2SO_4/t), carbonaceous BIF (ANC of 134-333 kg H_2SO_4/t) and Dolerites (ANC of 63-92 kg H_2SO_4/t) can offer some readily available neutralising potential. In most cases the risk assessment score for neutralising potential for Pilbara deposits is low.

4.1.2 Incipient AMD risk

Since AMD may take many years to manifest depending on the aridity of the climate and the host rock neutralising potential the age of the operation provides important information on the likelihood of AMD. New operations or a significant change to an existing operation (such as the recent initiation of mining below the water table) will be assigned the highest score.

4.1.3 Scale of disturbance

There is a greater potential for a large contaminant flux into the environment for larger masses of material exposed and therefore a higher score is given to mine sites with a large mass of mineral waste or a large disturbed footprints associated with waste disposal or open pits.

4.1.4 Transportation pathways

The ratio of precipitation to evapotranspiration is used as a proxy for the amount of water that is available to transport sulfide oxidation products from their point of production to the down gradient receiving environment. Within the Pilbara region the mean annual rainfall is typically 375 mm and annual evaporation varies from 3,000 to 3,600 mm. In arid climates such as the Pilbara the annual precipitation is much lower than the potential evapotranspiration rates. Most mines in the Pilbara have a precipitation to evaporation ratio of 1/10 to 1/3. To account for greater water availability and potential for contamination migration, ore bodies located below the water table are assigned a higher score than ore bodies located above the water table (Table 2).

Average local precipitation divided by areal potential evapotranspiration	**Existing operations**	**Exploration / Development**
< 1/10 ratio: mining above the water table exclusively	0	0
< 1/10 ratio: mining below the water table in an aquitard or an isolated aquifer	1	2
< 1/10 ratio: mining below the water table in a rock mass that is connected to a regionally significant aquifer	2	3
1/10 to 1/3 ratio: mining above the water table exclusively	1	2
1/10 to 1/3 ratio: mining below the water table in an aquitard or an isolated local aquifer	2	3
1/10 to 1/3 ratio: mining below the water table in a rock mass that is connected to a regionally significant aquifer	3	5
1/3 to 1/2 ratio	3	5
1/2 to 1.5/1 ratio	6	8
> 1.5/1 ratio	7	10

Table 2. Hazard scores based on the precipitation and potential evapotranspiration for the deposit.

4.1.5 Sensitivity of the receiving environment

The environmental sensitivity is assessed by assigning a score for the proximity to perennial and ephemeral water bodies, the buffering capacity of the receiving water and the proximity to protected or permanently inhabited areas.

4.2 Stage 2: technical AMD and geochemical risk assessment report

The identification of potential AMD issues at the exploration and feasibility phases is critical, as these mine planning phases are often linked with community consultation, environmental impact assessment and regulatory approvals. During feasibility studies for new mine sites there is a requirement in the Mineral Waste Management Plan for a detailed AMD and geochemical risk assessment report to be completed for those sites that scored moderate or high in the preliminary AMD Hazard Score. This report assesses the following information:

4.2.1 Background information

Background information on the sites geology, climate, hydrogeology, surface water and surrounding environment are necessary to understand the risk and potential impacts.

4.2.2 Sulfur distribution (drill hole data interrogation)

The total sulfur concentration is measured in most drill hole assays for a deposit and this data can be interrogated to assess the AMD risk.

4.2.2.1 Observed pyrite

The number of pyrite observations in each stratigraphic unit can be useful for assessment of risk, however this assessment can not be used alone due to the difficulty in observing pyrite in some samples depending on drilling method or the nature of the material.

4.2.2.2 Total sulfur analysis

Extensive geochemical, Acid Base Accounting (ABA) and Net Acid Generation (NAG) test characterisation work has found that a total sulfur concentration of 0.1% is the most appropriate boundary between non acid forming and potentially acid forming black shale. For other lithologies such as BIF and Detritals a 0.3% total sulfur concentration is the most appropriate boundary. However, these boundaries need to be re-confirmed for each new deposit to ensure they are appropriate.

The number of samples in each lithology with total sulfur concentrations exceeding 0.1% or 0.3% is evaluated to identify high risk lithologies. Selective management of some higher sulfur rock masses may not be needed in some circumstances depending on the geology and overall percentage of sulfur in the material to be disturbed by mining. It may also be difficult to define mineable units of some lithologies with low elevated sulfur percentages that are scattered within the lithology.

It is useful to look at the spatial distribution of elevated sulfur material within the pit shell using three dimensional software (Fig. 5). Occasionally elevated total sulfur concentration can be found within metres of the surface and in these cases it is likely that the sulfur represents sulfates rather than sulfides. At some mine sites (not known at RTIO Pilbara mine sites) this could also be due to acid sulfate soils.

4.2.2.3 Drillhole sulfur analysis considering proposed pit shell

The previous analysis used all drill hole data for the deposit and does not account for those materials that ultimately fall within the pit shell. Therefore an analysis of sulfur values within the pit shell is also undertaken (Table 4). Occasionally high sulfur values are found near the deposit but this material will ultimately not be mined. The previous analysis can be used to identify this material and it is important to consider this for any pit shell changes or

Strand-tag group	Total samples assayed for S	Number of samples with S>0.1%	Number of samples with S>0.3%	Percentage of total samples with S>0.1%	Percentage of total samples with S>0.3%
CLA	568	2	0	0.35	0.00
CAL	704	3	2	0.43	0.28
DET waste	1,170	27	6	2.31	0.51
DET mineralised	526	2	0	0.38	0.00
DOR	53	2	0	3.77	0.00
WD waste	280	0	0	0.00	0.00
ANG waste	879	6	6	0.68	0.68
ANG mineralised	154	0	0	0.00	0.00
N2U BIF	78	1	1	1.28	1.28
N2L BIF	106	0	0	0.00	0.00
NE1 BIF	264	0	0	0.00	0.00
NEW mineralised	895	1	1	0.11	0.11
NEW HYD	200	0	0	0.00	0.00
MAC BIF	192	12	8	6.25	4.17
MAC mineralised	68	1	0	1.47	0.00
MAC HYD	77	5	0	6.49	0.00
NAM BIF	59	8	1	13.56	1.69
UNKNOWN	1	0	0	0.00	0.00
Total number of samples assayed		6,274	6,274		
Total number of samples with S>0.1%/0.3%		70	25		
Percentage of total with S>0.1%/0.3%		1.12	0.4		
Total number of waste samples		4,353	4,353		
Total number of waste samples with S>0.1%/0.3%		61	24		
Percentage of total waste samples with S>0.1%/0.3%		1.40	0.55		

Table 3. An example of total sulfur analysis for a deposit.

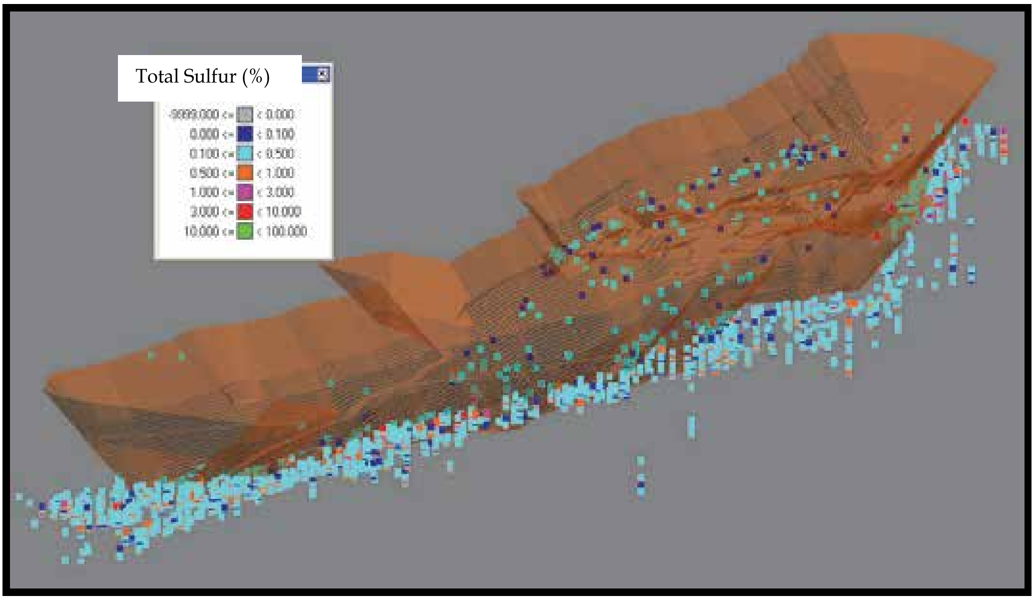

Fig. 5. An example of the spatial distribution of total sulfur (≥ 0.1%) in drill hole composites and the pit shell.

if there is any dewatering activity. During dewatering sulfides in the pit wall may become unsaturated and then once mining has finished and the water table recovers contaminants could be mobilised.

Total number of samples assayed for S within pit shell:	34,478
Number of samples with S>0.3% within pit shell:	97
Percentage of total with S>0.3% within pit shell:	0.28%
Total number of samples assayed for S within pit shell and BWT (580 mRL):	22,531
Number of samples with S>0.3% within pit shell and BWT:	92
Percentage of total with S>0.3% within pit shell and BWT:	0.41%

BWT= Below Water Table

Table 4. An example of the total sulfur value greater than 0.3%, within a deposit filtered using the proposed final pit design

4.2.3 Total sulfur analysis within the mining model

Sulfide risk categories have been created in the mining model so the tonnes of sulfidic material can be predicted. The total sulfur concentration also exists within the mining model and can be interrogated for sulfur risk by lithology and as a function of waste rock production over time (Table 5). Determining the tonnes of sulfidic material is important for assessing which lithologies present the greatest risk for AMD and for determining if there is adequate inert or neutralising material available for the proposed dump, co-disposal, encapsulation or cover designs.

	Sulfur bin (%)	ALLUVIALS	DETRITALS	ANG	N2L [illegible]	H2L	V2U [illegible]	HC1 [illegible]	[illegible]	NAC	UNKNOWN	Grand Total
High Grade	0.05	[illegible]		[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
	0.1	[illegible]		[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]		[illegible]
	0.15			[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]		[illegible]
	0.2				[illegible]	[illegible]	[illegible]	[illegible]	[illegible]			[illegible]
	0.25					[illegible]			[illegible]			[illegible]
	0.3					[illegible]			[illegible]			[illegible]
High Grade Total		378 354		13,330,686 [illegible]	[illegible]	112,703,330 [illegible]	137,315,381 [illegible]	60,582,776 [illegible]	20,480,663 [illegible]	925,735 [illegible]	15,615	[illegible]
Low Grade	0.05	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
	0.1	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
	0.15		[illegible]	[illegible]	320 177 [illegible]	455,331 [illegible]	[illegible]	[illegible]	[illegible]	[illegible]		2 132 888 [illegible]
	0.2		[illegible]		[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]		[illegible]
	0.25					[illegible]		[illegible]	[illegible]			[illegible]
	0.3					[illegible]		[illegible]	[illegible]			[illegible]
	0.35							[illegible]	[illegible]			[illegible]
	0.4							[illegible]				[illegible]
Low Grade Total		[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	27,743,214 [illegible]	[illegible]	[illegible]	[illegible]	104,939 [illegible]	[illegible]
Waste	0.05	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
	0.1	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
	0.15	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	12 422 470 [illegible]	[illegible]	[illegible]
	0.2	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]		[illegible]
	0.25			[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]		[illegible]
	0.3			[illegible]				[illegible]	[illegible]	[illegible]		[illegible]
	0.35			[illegible]		[illegible]		[illegible]	[illegible]			[illegible]
	0.4			[illegible]		[illegible]		[illegible]				[illegible]
	0.45			[illegible]								[illegible]
	0.5			[illegible]								[illegible]
	100								[illegible]			[illegible]
Total Waste		[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
Grand Total		322 644,495 [illegible]	13,353,527 [illegible]	[illegible]	[illegible]	[illegible]	227,375,481 [illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]

Table 5. An example of estimated volumes of material predicted to be mined at a deposit (for all wet and dry material, in tonnes)

4.2.4 Potential sulfide exposures on the final pit walls

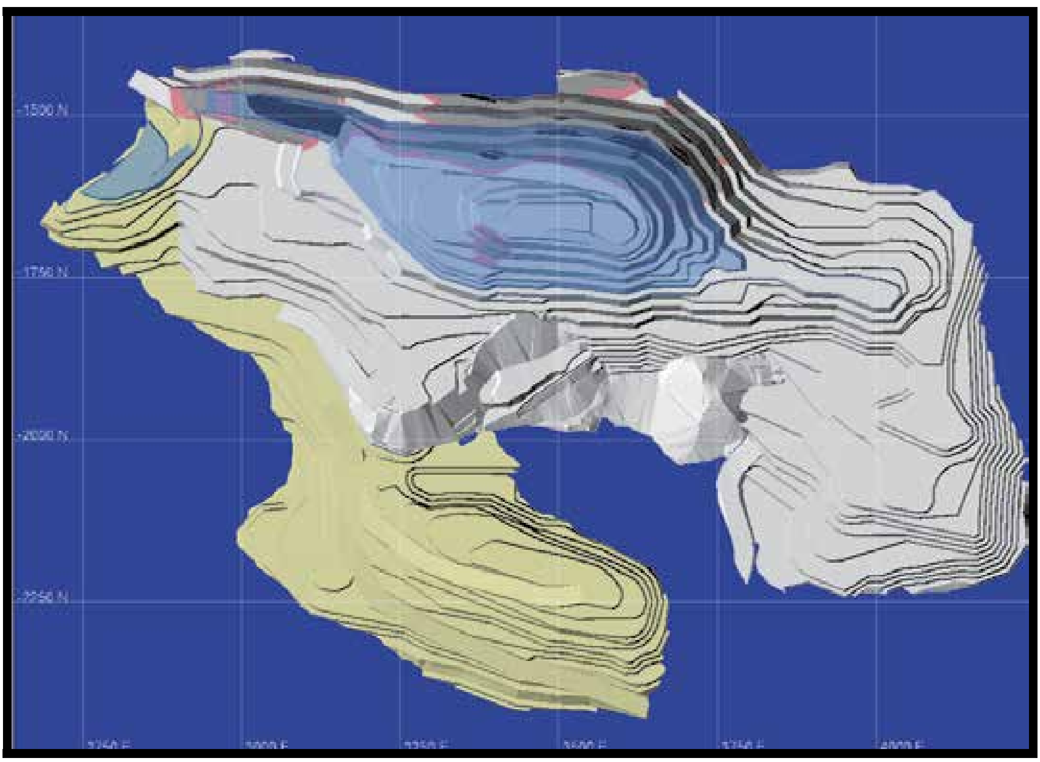

Fig. 6. An example of surface exposures of PAF material relative to the pit void catchment (light grey, where yellow represents the area which is unlikely to contribute to surface water runoff). Oxidised material = pink, low risk = dark grey, high risk = black and blue represents the pre-mining water table.

Predicting the surface area and location of Potentially Acid Forming (PAF) material at mine closure provides information on the risk of an acidic pit lake developing at mine closure (Fig. 6). This information can be used to dictate necessary backfill levels, surface water diversions or be used in final void water quality modelling studies to predict the evolving water quality of the pit lake. Predicting the surface area and location of PAF material year by year can also be useful in regard to predicting the quality of the surface water runoff generated during mining. This information could be used to limit PAF exposures during typically high rainfall periods and thereby reduce the amount of potentially contaminated water requiring treatment.

4.2.5 Acid base accounting test work results

Recognised ABA and NAG analytical techniques provide confirmatory information on typical Non Acid Forming (NAF)/PAF cutoffs based on total sulfur (AMIRA 2002; DoITR 2007; Gard Guide 2009; Price 2009). The low capacity to generate acidity can also be identified. Sometimes it can be difficult to determine if a sample is NAF or PAF and an uncertain classification can be assigned. These tests can also provide useful information on the neutralising capacity of a sample, the amount of potential acidity and its rate of release, other contaminants that are enriched and could mobilise into water and intrinsic oxidation rates. RTIO also undertake additional tests to determine the reactivity of the material with nitrogen based explsoives. The premature detonation of explosives with nitrogen based explosives is a safety risk for some materials and inhibited explosives are used when necessary to reduce this risk.

4.2.6 Chemical enrichment

4.2.6.1 Solid enrichment

Trace element data (Al, As, Ca, Cl, Co, Cr, Cu, Fe, Pb, Mg, Mn, Ni, P, K, S, Si, Na, Sr, Ti, V, Zn and Zr) is routinely collected from drill hole samples and is analysed as part of the AMD and geochemical risk assessment report to determine chemical enrichment. The extent of enrichment is reported as the Geochemical Abundance Index (GAI), which relates the actual concentration with median crustal abundance (Bowen 1979) on a log 2 scale. The GAI is expressed in integer increments where a GAI of 0 indicates the element is present at a concentration similar to, or less than, median crustal abundance and a GAI of 6 indicates approximately a 100 fold enrichment above median crustal abundance. As a general rule, a GAI of 3 (about a ten fold enrichment) or greater signifies enrichment that warrants further examination.

In addition, to this detailed look at assay information in the drill hole database, chemical enrichment is determined for each major lithology type during major drilling campaigns. The GAI is calculated for each lithology and additional less commonly enriched elements are also periodically analysed (ie. Ag, B, Be, Cd, F, Hg, Mo, Sb, Se, Th and U). A table of trigger values has been generated within the Mineral Waste Management Plan and this table can be used for quick comparison of concentrations (rather than calculating the GAI each time).

4.2.6.2 Liquid extracts

Solid enrichment of an element does not necessarily pose environmental risks unless the element is also bio-available and/or can be mobilised into surface and groundwater. A

Analyte	mg/kg or ppm	%
Ag	0.59	
As	13	
B	85	
Ba	4,243	0.4
Be	22.06	
C	20,000	2
Cd	0.93	
Cl	1,103	
Co	170	
Cr	849	
Cu	424	
F	8,061	0.8
Hg	0.42	
Mn	8,061	0.8
Mo	10.2	
Ni	679	
P	8,485	
Pb	119	
S	1,000	0.1
Sb	1.70	
Se	0.42	
Sn	19	
Sr	3,140	0.3
Th	102	
U	20	
V	1,358	
Zn	636	

Table 6. Trigger values based on the median crustal abundance. [1]

liquid extract test is undertaken to provide a quick indication of contaminant mobility. A solid and liquid water extract (1:2 ratio respectively) is thoroughly mixed and left overnight before the liquor is siphoned off and then the pH and Electrical Conductivity (EC) is measured. The liquor is then filtered (through a 45 µm filter), acidified and analysed. The average concentration for each element from each lithology is then compared against background concentrations, ANZECC and ARMCANZ (2000) stock water guidelines or NHMRC (2004) Australian drinking water guidelines depending on the likely end water use. The liquid extracts are a quick indication of the:

- Leachability of metals under the prescribed laboratory conditions (crushed samples, pure water as a leachant and a known water-to-rock ratio); and
- The condition of the sample with respect to weathering (ie if the sample is 'fresh', or if it is PAF but has not yet acidified, the test may not necessarily identify all the metals of concern in the longer term). However, while these laboratory tests may be used to infer which contaminants might be released from the materials under laboratory conditions, they do not necessarily reflect the metal concentrations that may occur in leachates generated in the field.

The overall objective of the geochemical analysis is to provide a quick first pass test to determine whether the waste material to be mined is inert. If geochemical test work indicates that the waste lithology may not be inert then further analysis such as column leach or humidity cell experiments are undertaken. These kinetic tests are run over many months or years.

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[1] Triggers were derived from the median crustal abundance (Bowen 1979). The values are equivalent to a GAI of 2.5 and when rounded up 3 (i.e. $10^{(3x\log(2))x1.5x(\text{crustal abundance})}$). This is equivalent to an 8.5 times increase above the median crustal abundance.

4.3 Stage 3: detailed AMD hazard score

The technical AMD and geochemical risk assessment report provides sufficient information to complete the detailed AMD Hazard Score Assessment. The RTIO AMD Hazard Score was developed to ensure a consistent assignment of risk for each deposit and operation at RTIO's Pilbara operations.

2. Detailed Assessment (Pre Feasibility/ Feasibility/Mining)

This assessment should be completed by an AMD expert

Pit	**Example site - BWT**		
F. Geochemical Hazard (Interrogate the drill hole database)			
Geochemical Summary			
Number of total sulfur concentrations collected	87,341		
Lithologies assayed	All major material types within the pit shell		
Likely PAF materials in bulk	**Nil**	If relevant, list lithologies	
	Comments	**Example site - BWT**	**Other RTIO mine sites within similar lithology**
Number of acid base accounting (ABA) samples	Due to lack of sulfides found no ABA could be undertaken	0	38
Number of column leach experiments	Due to lack of sulfides found no ABA could be undertaken	0	3
Score			
	Select Relevant Option Below	**Score**	**Option Details**
Waste sulfur risk	Total number of waste samples with S>0.1% is less than 3%	0	For total drillhole samples, 0.78%; for waste drillhole samples, 0.71%
Ore grade sulfur risk	Total number of ore grade samples with S>0.1% is less than 3%	0	
Spatial distribution of sulfur	Sulfur scattered throughout the pit and through numerous lithologies	3	Unlikely that sulfur represents sulfides
Chemical enrichment	Enrichments of contaminants that are unlikely to mobilise into groundwater	1	As, Fe, Sn enriched but unlikely to be mobile
G. Mine Planning Hazard			
	Select Relevant Option Below	**Score**	**Option Details**
PAF material management	No special waste management needed	0	
Bulk NPR (Mass of neutralising material x mean ANC) / (Percent of lithology greater than 0.1% x tonnes of lithology x mean sulfur concentration for all data greater than 0.1 x 30.6 + repeat for each PAF lithologies)	>3	0	estimated
PAF rock mass disturbed or exposed (waste tonnes with S>0.1%)/(total tonnes of waste)*100	< 3% of the total disturbed mass	0	No PAF material expected
Pit backfilling	Pit will be backfilled to above the post mining water table but below ground surface	2	Proposed
H. Water Management Hazard			
	Select Relevant Option Below	**Score**	**Option Details**
Dewatering volume	80-160 ML/day	2	Peak max. 100 ML/day
Surface water	Creek flow	7	
Water treatment during Operation	No water treatment or special management for AMD needed	0	
Final void management	No PAF rock exposures likely on final pit shell	0	
Combined Hazard Assessment			
Preliminary Assessment Score	49		
Detailed Assessment Score	15		
Combined Hazard Score	27		
Risk Ranking	LOW		

Fig. 7. Example of the use of the detailed AMD Hazard score to assess a site.

The preliminary AMD Hazard Score is relevant during order of magnitude or exploration studies where information is lacking however during pre-feasibility, feasibility, development or mining of a deposit a more refined, defensible and repeatable hazard assessment is required. The hazard assessment should lead to a consistent assignment of risk so that all personnel involved in project development understand the implications of each risk rating.

The ranking system outlined in the following section is designed to identify those orebodies, open pits and waste rock dumps which, though they may contain small amounts of PAF material, are unlikely to pose a risk to water quality or revegetation programs. No special waste or water management above that already required for inert materials would be required for these low risk sites. Conversely a high risk site could generate widespread AMD and environmental impacts without special management of waste rock and water during operation. Acidic pit lake formation would be near certain without extensive backfilling at closure. To control the potential AMD impacts from a high risk site, strategic changes to the life of mine plan would likely be justified. PAF materials would also probably require special management at moderate risk sites, but given sulfur contents and material balances, the management could be easily addressed at an operational/tactical rather than a strategic level.

The RTIO detailed AMD Hazard Score is specific for the Pilbara operations and can be used to compare the AMD risk of different operations against each other (Fig. 7). However, because it is specific to iron ore deposits in the Pilbara region, the hazard score is conservative and is likely to over-estimate the risk when compared against porphyry copper or some coal deposits. A summary of the different categories within the detailed AMD Hazard Score are discussed in the following sections:

4.3.1 Geochemical hazard

An assessment of the total sulfur content in waste and ore and the overall spatial distribution of sulfur in the deposit are used to provide a detailed geochemical hazard score. All data for this analysis should be derived from the drill hole database.

4.3.1.1 Waste sulfur risk

Waste sulfur risk	**Score**
Total number of waste samples with S>0.1% is less than 3%	0
Total number of waste samples with S>0.1% is between 3% and 10%, less than 0.5% of samples have S>0.3%	2
Total number of waste samples with S>0.1% is between 3% and 10%	7
Total number of waste samples with S>0.1% is greater than 10%	10

Table 7. Scores assigned to waste sulfur risk.

All total sulfur measurements for waste rock within the deposit or pit should be used to determine the waste sulfur risk. It is conservatively assumed that all total sulfur measurements represent sulfide minerals (i.e. pyrite) however it is likely in some deposits that sulfur near the surface is actually in the form of sulfate minerals (i.e. gypsum, alunite, schwertmannite, jarosite).

The number of samples per waste lithology with a total sulfur concentration greater than 0.1% can be calculated using strand/tag or geozone information however if this data has not been populated then stratigraphy logging can also be used. This value should be compared against the total number of waste samples assayed to determine the relative risk (Table 7).

4.3.1.2 Ore grade sulfur risk

Using a similar methodology to Section 4.3.1.1 the number of ore grade samples with total sulfur measurements greater than 0.1% should be compared against the total number of ore-grade samples to determine the relative risk (Table 8). Scores are lower for the sulfur characterisation of ore compared to waste due to most ore being transported away from the mine site.

Ore grade sulfur risk	**Score**
Ore grade material will not be stockpiled	0
Total number of ore grade samples with S>0.1% is less than 3%	0
Total number of ore grade samples with S>0.1% is between 3% and 10% but less than 0.5% of the samples have S>0.3%	2
Total number of ore grade samples with S>0.1% is between 3% and 10%	4
Total number of ore grade samples with S>0.1% is greater than 10%	5

Table 8. Scores assigned to ore grade sulfur risk.

4.3.1.3 Spatial distribution of sulphur

Spatial distribution of sulfur	**Score**
Sulfur scattered throughout the pit and through numerous lithologies	3
Sulfur concentrated within one or two lithologies (i.e. MCS and FWZ)	5

Table 9. Scores assigned to spatial distribution of sulfur.

High sulfide sulfur zones that are scattered throughout the deposit will be difficult to selectively manage compared to high sulfur zones confined to one or two lithologies. Overall sulfide oxidation within waste dumps that group all high sulfur material together will generally be lower than if high sulfur material is broadly intermixed with inert material. This is particularly true if the high sulfur material is encapsulated or covered with inert material. However, high sulfur material scattered throughout the deposit is also likely to be diluted as it is mined and it is possible that any neutralisation potential in the country rock or groundwater may have capacity to buffer the acidity released compared to the acidity released from a single large mass of high sulfur rock concentrated in one location. Typically within RTIO Pilbara operations the sulfur scattered throughout the deposit has low total sulfur concentrations (i.e. < 0.3%) and therefore this risk is deemed lower than that of sulfur concentrated within one or two lithologies (Table 9).

4.3.1.4 Chemical enrichment

The mean concentration for each element measured in the lithology should be compared to the average crustal abundance to determine if there is significant enrichment (Section 4.2.6). In some cases further test work (i.e. liquid extracts or kinetic leach experiments) may be necessary to assess the overall risk of an enriched element becoming mobile within surface water or groundwater aquifers (Table 10).

Chemical enrichment	**Score**
No enrichment of contaminants	0
Enrichments of contaminants that are unlikely to mobilise into groundwater	1
Enrichments of contaminants that are likely to mobile into groundwater	5

Table 10. Scores assigned to chemical enrichment risk.

4.3.2 Mine planning hazard

The mine planning hazard score is determined by analysing the mining model for the quantity of PAF material as delineated by a sulfide risk variable, the relative tonnes of neutralising material, and also considers the tonnes of material with elevated sulfur grades. Waste dump plans should also be assessed for risk to the receiving environment.

PAF material management

PAF waste dumps located in pit are more secure than disposal in above ground rock dumps (Table 11). In pit disposal is the preferred disposal location due to:

- Reduced risk of erosion exposing sulfides in the long term;
- Inhibiting convective oxygen transport because the waste is surrounded by relatively impermeable rock walls;
- Reduced footprint of the waste disposal facilities;
- Reduced volume of inert or net neutralising waste needed to encapsulate the sulfides; and
- The formation of acidic or hyper-saline pit lakes may be prevented if the pit can be filled to above the post-mining water table.

PAF material management	**Score**
No special waste management needed	0
PAF waste dumps will be in-pit	2
PAF waste dumps will be in pit and out of pit	4
PAF waste dumps will be out of pit	5

Table 11. Scores assigned to PAF material management.

4.3.2.2 Bulk neutralisation potential ratio

The Neutralisation Potential Ratio (NPR) can be used to provide a quick bulk assessment of the likelihood of alkalinity within other lithologies buffering any acidity produced (Table

12). It is unlikely that neutralisation will be 100% effective and geochemical characterisation may be necessary to confirm the characteristics of material at the site. The bulk NPR can be calculated by:
[mass of neutralising material x mean ANC]
[mass of acid producing material x mean potential acidity]
The bottom line of the equation is calculated by the sum of all acid producing lithologies:
[Lithology 1: percent of lithology with S greater than 0.1% x total tonnes of lithology x mean sulfur concentration of lithology for all samples with sulfur assay values greater than 0.1 x 30.6]
+
[Lithology 2: percent of lithology with S greater than 0.1% x total tonnes of lithology x mean sulfur concentration of lithology for all samples with sulfur assay values greater than 0.1 x 30.6]
+
[Lithology 3 etc]

Bulk NPR of entire rock mass to be disturbed or exposed	Score
<1	5
1 to 3	3
>3	0

Table 12. Scores assigned to NPR.

4.3.2.3 PAF rock mass disturbed or exposed

The tonnes of PAF rock mass disturbed can be calculated by extracting the tonnes of material with S>0.1% in the mining model or from sulfide risk variables that have been added to the mining model. If the sulfide risk variable is available then this should be used in preference to evaluate the total tonnes of material with S>0.1%. This analysis provides a more detailed assessment for the scale of disturbance which was addressed in the preliminary assessment (Table 13).

PAF rock mass disturbed or exposed	Score
< 3% of the total disturbed mass	0
3 to 10% of the total disturbed mass	5
> 10% of the total disturbed mass	10

Table 13. Scores assigned to PAF rock mass disturbed or exposed.

4.3.2.4 Pit backfilling

A pit that is backfilled when the mine is closed is likely to have a lower risk of AMD generation compared to an open pit (Table 14). Covering sulfide exposures will also reduce the risk of AMD.

4.3.3 Water management hazard

The water management hazard score is derived from an assessment of likely water discharge volumes and quality. The final void water quality is also considered as this can contribute significantly to the mine closure cost.

Pit backfilling	Score
Pit will not be backfilled	5
Pit will be backfilled below the post mining water table	4
Pit will be backfilled to above the post mining water table but below ground surface	2
Waste will be tipped over black shale exposures	2
Pit will be backfilled to ground level	0

Table 14. Scores assigned to pit backfilling scenarios.

4.3.3.1 Dewatering volume

Dewatering of mine voids is required to provide access to below watertable ore and to reduce geotechnical risk of slope failures. On mine closure there is potential for AMD generation as sulfides are rewetted by the recovering water table. A more detailed investigation would be required to quantify this risk (for example investigating the distribution of sulfur in the pit wall). A large dewatering campaign could also be more of a problem if the groundwater became acidic in the future owing to leaching of acidic material from pit walls (Table 15).

Water discharge	Score
No releases of water	0
0 to 80 ML/day	1
80-160 ML/day	2
> 160 ML/day	3

Table 15. Scores assigned to water discharge.

4.3.3.2 Surface water management

Surface water is likely to more significantly contribute to AMD generation than groundwater within the Pilbara. Therefore, the combined scores of an assessment of the pit surface area and the surface water catchment are greater than the score for dewatering discharge in Table 15 (Table 16). Surface water management plans and/or consultation with site personnel or RTIO hydrologists will be necessary to determine the risk of increased surface water runoff from the catchment above a pit or from a creek that has not been diverted around a pit.

Surface water	Score
Isolated pit	0
Catchment area above the pit	5
Creek flow	7

Table 16. Surface water assessment of the pit.

4.3.3.3 Water treatment during operation

Water requiring treatment during operation may also require treatment on mine closure. The cost during operation and mine closure may be significant (Table 17).

Water treatment during operation	Score
No water treatment or special management for AMD needed	0
Water treatment or special water management may be needed during operation	3
Water treatment or special management will be needed during operation	5

Table 17. Scores assigned to water treatment during operations.

4.3.3.4 Final void management

Large exposures of elevated sulfur material on the pit wall are more likely to generate an acidic pit lake on mine closure. Acidic voids are unlikely to be acceptable to the regulators on mine closure and therefore ongoing treatment or backfilling could be required (Table 18). Final exposures on the ultimate pit wall can be calculated using the final pit shell and sulfide risk variables or geology strands. The detailed AMD and geochemical risk assessment report should also investigate the position of this material relative to the post-mining water table (if available) (Fig. 6).

Final void management	Score
No PAF rock exposures likely on final pit shell	0
Less than 3% PAF exposed	2
3% to 10% PAF exposed	7
Greater than 10% PAF exposed	10

Table 18. Scores assigned to final void management.

4.3.4 Combined hazard assessment

The RTIO detailed AMD Hazard Score has been calibrated with data from the existing AMD and geochemical risk assessment reports, known risks at several mine sites and judgement of AMD experts.

The combined AMD hazard score is derived by adding the individual scores relating to the preliminary assessment, detailed geochemistry, mine planning and water management. A score of 30 or less receives a low AMD hazard ranking. These sites are the least likely to generate significant AMD or cause significant metals loading into the environment. A score between 30 and 50 receives a moderate hazard ranking. These sites are more likely to generate either significant AMD or circum-neutral pH contact waters with elevated salinity and/or metals content. A score of 51 to 65 receives a high AMD hazard ranking, and a score of 66 or higher receives a very high ranking. These sites pose a significant environmental, financial and/or reputational risk because of their potential to generate large AMD fluxes.

4.4 Stage 4: AMD risk assessment of management strategies

The final stage in the risk assessment process involves analysis of all possible scenarios, causes and potential impacts. An inherent risk is assigned based on consequence and likelihood. Inherent risk provides an indication of the "true" risk of the impact occurring when there are no controls in place to mitigate the risk. To score inherent risk it is assumed that the impact will occur and therefore the probability descriptors of almost certain, likely or possible should be used and unlikely or rare can not be used.

Some examples of inherent risks from AMD include:

- Sulfidic material within waste dumps generates AMD in surface and groundwater;
- Spontaneous combustion or convective gas transport within the dump causes dump instability;
- The final pit lake that develops once mining ceases is polluting, impacting local groundwater and fauna;
- Dewatered water develops into AMD and impacts on flora and fauna if it is disposed of within a creek;
- Sulfidic exposures on the pit wall react with rainwater to generate AMD within the pit causing health and environmental impacts; and
- Re-establishment of water table post mining causes dissolution of efflorescent salts resulting in increasing contaminant concentrations in groundwater.

A current risk is then assigned based on the implementation of controls and management measures. If necessary the residual risk is also addressed. Controls can be physical, procedural and behavioural. Some examples of controls that could be implemented to reduce risk include:

- Encapsulation of sulfidic material within inert material;
- Placement of covers over sulfidic material ie. store and release, shedding, alkalinity;
- Appropriate co-disposal of material with neutralisation potential;
- Acid water treatment or containment systems;
- Bunding to separate inert water from AMD;
- Training;
- Management plans and auditing for compliance against the plans; and
- Pit backfilling to above the post-mining water table or to cover PAF material exposed on the pit wall.

5. Conclusions

One of the key challenges facing the mining industry is the management of AMD, to minimise risks to human health and the environment. A crucial step in leading practice management of AMD is to assess the risk as early as possible, so that appropriate pro-active management strategies can be selected and implemented. This includes assessment of environmental, human health, commercial and reputation risks. RTIO have developed a four stage risk assessment process to thoroughly assess the risk of AMD:

1. Preliminary AMD Hazard Score
2. Technical AMD and geochemical risk assessment report
3. Detailed AMD Hazard Score
4. AMD risk assessment of management strategies

Progressively more knowledge is required through each of the stages to analyse the risk. All stages can be completed prior to mining and this allows the AMD risk to be fully evaluated before considerable investment or works have occurred. The upfront identification of risk means that options such as avoidance and appropriate management strategies can be appropriately explored. Effort is focused on pro-active prevention or minimisation rather than control or treatment whenever possible.

The quantitative AMD Hazard score means that a consistent assignment of risk is assigned to each deposit and operation. It is accompanied by a technical risk assessment completed by an AMD expert to ensure the quantitative score is reasonable. Finally the risk to human health and environment is assessed individually and then reassessed after appropriate management strategies have been implemented.

6. Acknowledgements

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A Study of Elevated Temperatures on the Strength Properties of LCD Glass Powder Cement Mortars

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1. Introduction

The rapid increases in population, urbanization, and economic development, have been accompanied by an increase in the accidental fire risk. The fire redundancy of buildings can reduce the injury and damage, enhance the safety of residents, and increase the reusability of buildings. These are the prevailing concepts behind the development of fire proof buildings (Fang, 2006). The advancements in optoelectronic technology, software technology, and other high-tech production have made Taiwan a "green silicon island" over the global high-tech service and manufacturing industries. Unfortunately, these developments have also generated a considerable amount of industrial waste that, if handled improperly, will cause severe environmental damages. Recently, researchers have suggested that these industrial wastes are of high potential to be recycled, to generate economic benefits, and to reduce the dependency on national resources (Cheng, 2002). Rapid industrial development and high life standard have both increased the amount of waste glass, of which only a limited fraction is properly recycled and reused (Park et al., 2004; Mohamad, 2006). Liquid crystal products such as LCD screens and mobile phone panels have become increasingly popular in recent years. Taiwan's TFT-LCD panel manufacturing products have been ranked as the top 1st over the world, which account for 39.2% of the entire global output. The LCD waste glass generated from the manufacturing process is approximately 12,000 tons per year (Cheng, 2002; Fang, 2006). How to use LCD glass waste in producing concrete has therefore, become a highly attractive issue in Taiwan.

Glass waste is considered as ecologically friendly and non-toxic, with qualified physical properties and a simple chemical composition. For example, soda-lime glass consists of approximately 73% of $Si0_2$, 13% of Na_20 and 10% of CaO (Shi and Zheng, 2007.). This renders most glass wastes environmentally friendly as a recyclable material (Cheng and Chiang, 2003). The term "glass" comprises several chemical varieties, including binary alkali-silicate glass, boro-silicate glass, and ternary soda-lime silicate glass (Shayan and Xu, 2006). One solution to properly recycle these glass wastes is suggested by grinding the material into fine glass powder (GLP), and incorporating them into concrete as a pozzolanic agent. Laboratory experiments have shown that fine GLP is capable of suppressing the alkali reactivity present in coarser glass aggregates and naturally obtained reactive aggregates. In addition, finer glass powders are beneficial to the pozzolanic reactions in concrete. It was reported that a replacing amount of 30% cement by glass powders in some mixes has shown to provide satisfactory mechanical strengths (Shayan and Xu, 2004).

Most reused glass is produced through the re-melting process. Therefore, not all waste glass is suitable for producing recycle glass, particularly for those beverage bottles. This is because they are mostly contaminated with paper and other undesired substances. For quality and security purposes, the outlets of waste glass must be properly identified, especially when using in the construction industry (Lin, 2006). Previous literature related to the functionality of waste glass in concrete production has focused on its application as a substituent for cement. Other successful examples of waste glass recycling projects include using recycled glass as a cullet in glass production, a raw material for the production of abrasives and fiberglass, an aggregate substituent in concrete (as a pozzolanic additive), an agent in sand-blasting, road beds, pavement and parking lots, a raw material for the production of glass pellets or beads used in the reflective paint of highways, and a fractionators for lighting matches and firing ammunition (Poutos et al.2008; Zainab and Enas,2009). Previous investigation shows that the compressive, flexural, indirect tensile strengths and Schmidt hardness of concrete would decrease as the content of waste glass aggregate increases, particularly when the content exceeds 20% (Bashar and Ghassan, 2008). Although the influence on the mechanical properties of concrete is not thoroughly characterized, the employment of recycled glass is still rapidly emerging, and can widely be found in many industries such as asphalt concrete (glasphalt), normal concrete, back-filling, sub-base, tiles, masonry blocks, paving blocks and other decorative employments (Jin et al.,2000; Dyer and Dhir, 2001; Xie et al., 2003; Topcu and Canbaz, 2004; Park et al., 2004).

Using waste glass as a finely ground mineral additive (FGMA) in cement is another potential application (Bashar and Ghassan, 2008). The primary concern regarding the use of glass in concrete is the chemical reaction that takes place between the silica-rich glass particles and the alkali environments in the concrete pores (alkali-silica reaction). This reaction is detrimental to the stability of concrete properties unless appropriate precautions are taken to minimize this negative effect. Preventative actions include the incorporation of suitable pozzolanic materials such as fly ash, ground blast furnace slag (GBFS), or met kaolin in the concrete mix (Al-Mutairi et al., 2004). Nevertheless, Shayan and Xu have found that a 30% content amount of glass powder could be incorporated as the fine aggregate or cement replacement in concrete without causing any long-term detrimental effects (Shayan and Xu, 2004). Other results have also revealed that there is an increase in the concrete compressive strength if waste glass of very fine grade is added (Federico and Chidiac, 2009). Glass contains large quantities of silicon and calcium, which is very similar to Portland material in nature. Its physical properties such as density, compressive strength, modulus of elasticity, thermal coefficient of expansion, and coefficient of heat conduction are also very close to those of concrete (Topcu and Canbaz, 2004). Previous research results have shown that the fluidity, air content, and unit weight of concrete would increase if glass sand is employed as the fine aggregate substituent (Zeng, 2005). In addition, researchers have reported that the compressive strength, flexural strength, and cleavage strength of concrete would increase with the amount of glass powder inclusion, while the optimum adding fraction is about 20% (Zeng, 2005; Wang et al., 2007). Hence, Chi Sing Lam et al. suggested that glass sand can be purposely used to economically design the strength, to effectively decrease the porosity, and to enhance the durability, ultrasonic velocity, and resistance to acid, salt, alkali, and chloride ion electric osmosis of concrete (Wang, 2010). In recent years, recycling waste LCD glass has become an important issue in Taiwan (Wang, 2010). It has been reported that controlled low strength materials (CLSM) containing waste LCD glass would meet many engineering property requirements including the strength, high fluidity, high permeability, and low electrical resistivity. All these measures would thus usher in the innovative application of waste glass (Hsu, 2009).

The contamination, residue, and organic content of recycled waste glass sand may be disadvantageous for construction application because these will result voids generated within the concrete micro-structures, and consequently degrade the physical properties over time (Konstantin et al., 2007). However, observation from scanning election microscope (SEM) has revealed a visible densification around the glass grains, due to their partial hydration and the formation of additional C-S-H gel. The SEM investigation has shown that the main difference between glass cement and Portland cement pastes was the shrinkage in the CH crystal size and amount. This is caused by the glass grains involved in the pozzolanic reaction, leading to the consumption of CH crystals (Konstantin et al., 2007). Very recently, researchers have proposed that using waste glass from liquid crystal panels to replace fine aggregate and cement in concrete is a pioneering step for waste recycling technology in Taiwan (Wang and Huang, 2010a, 2010b; Wang, 2009a, 2009b,2011;Wang and Chen, 2008). When mixed at room temperature, the compressive strength of waste TFT-LCD glass cement mortar could achieve 211kgf/cm^2, while the specimens treated at elevated temperatures would behave even stronger. Although the alkali activators contain no sodium silicate, the compressive strength of TFT-LCD glass cement mortar can still be increased with adequate temperature treatment and mode of curing. All these are done for the purpose of reducing concrete production costs. Concrete made with LCD glass powder has a sharp aroma, but the inclusion of calcium hydroxide could eliminate the bad smell. Concrete slurry made with LCD glass powder has lower water permeability than that made with Portland cement, showing that glass cement mortar would generate a more compact micro-structure. The experimental results of alkali activators in waste glass cement slurry also indicate that glass sand could perform as well as the fine aggregate in forming the bonding agents, and could be used as the substituent for Portland cement (Ju, 2008). With all these promising outcomes presented, this study continues to address the influences of temperature on concrete strength, and the resistance of glass powder cement mortar to high temperature. We would demonstrate that the temperature resisting property of waste LCD glass cement mortar is a merit for enhancing the recycling value and the economic efficiency of waste LCD glass.

2. Experimental plan

2.1 Materials

This study used ASTM Type I Portland cement with the specific gravity of 3.15 and the Blaine fineness of 3519 cm^2/g. The corresponding chemical composition are SiO_2 (22.01 %), Al_2O_3 (5.57%), FeO_3 (3.44%), CaO (62.80%), K_2O (0.78%), Na_2O (0.40%), and MgO (2.59%) with trace amounts of TiO_2. The waste LCD glass was provided by Chi-Mei Industrial Corp., Taiwan. To achieve the uniformity of particle size, the TFT-LCD waste glass was crushed, grinded, and passed through a #8 sieve, respectively. The grinded particles were then dried using a Planetary Mill (Pulversette 4). Table 1 shows the corresponding results of toxic chemical leaching procedure (TCLP). The fine aggregates used for the mortar mixtures were obtained from Ligang River, which have also been approved following the ASTM C295 standards.

Component	As	Cd	Cr	Cr6+	Hg	Pb	Se
LCD glasses powder	0.022	ND	ND	ND	0.0077	0.281	ND
Regulatory	5.0	1.0	5.0	2.5	0.2	5.0	1.0
Remark	ND：Not detected						

Table 1. TCLP of LCD glass (mg/L)

2.2 Experimental variables and mixtures

The glass powder cement mortars used in this study were mixed at three different W/B ratios – 0.47, 0.59, and 0.71. The fineness values of the glass powder were 1500, 4500, and 6000 cm^2/g, and the replacement ratios were 0%, 10%, 20%, and 30% by weight, accordingly. The testing ages of the samples were 7 days, 28 days, 56 days, and 91 days. Elevated temperature at 105˚C, 580˚C, and 800˚C were treated onto the specimens, with the detail procedure described later in 2.3. It should be mentioned that the water content used in this study had included the water absorbed by sand aggregates. As shown in Table 2, the water amount for W/B ratios of 0.47, 0.59, and 0.71 were 265, 325, and 385 g, respectively.

NO.			Ceme nt	Glass	Sand	Water	Glass powder fineness (cm^2/g)
	G0		500	0			
	G1	F1	450	50			
	G2	F1	400	100			
	G3	F1	350	150			
47	G1	F4	450	50		265	1500
59	G2	F4	400	100	1375	325	4500
71	G3	F4	350	150		385	6000
	G1	F6	450	50			
	G2	F6	400	100			
	G3	F6	350	150			

Table 2. Mixture proportions of cement mortars

2.3 Experimental methods

For the fresh property examination, flow test (according to the CNS 1176 standard) and setting time test (according to the CNS 785 standard) were both conducted. Specimens are prepared with the geometry of 25×25×25 mm for compressive strength test (CNS 1232 standard) and 40×40×100mm for flexural strength test (CNS 1233 standard). According to ASTM C1012 standards, the anti-sulfate attack test was also performed with the mortar specimens cured for 7 days. After removed from the curing cabinet, the specimens were dried for 24 hours and then dipped with sulfates for another 24 hours; this was denoted as one cycle of sulfate corrosion attack. The weight loss of the specimens was measured and their appearance was simultaneously observed over the 5 cycles of corrosion attack. As for the high-temperature resistance test, compressive strengths of mortar specimens were investigated after several temperature treatments (105˚C, 580˚C, and 800˚C). Each temperature treatment consists of three steps: constantly increase to the target temperature within 2 hours, maintain the temperature for another 2 hours, and then lower the temperature back to normal in the last 2 hours. The glass powder morphology and microstructures of the mortar specimens were examined using a scanning electron microscope (SEM), JEOL JSM-6700F Japan. Glass powders were spread on a conductive double-edged adhesive tape that would then be attached to an SEM sample stud. Loose particles were properly dislodged with air blast. Representative photographs were taken after each sample was thoroughly observed.

3. Results and analysis

3.1 Chemical composition of waste LCD glass

Glass powders made from waste LCD glass consist of SiO_2 (62.48%), Al_2O_3 (16.76%), FeO_3 (9.41%), CaO (2.70%), K_2O (1.37%), Na_2O (0.64%), MgO (0.2%), and trace amounts of TiO_2, P_2O_6, and MnO. Table 1 presents the toxic chemical leaching procedure (TCLP) test results. As shown, the toxic contents of LCD glass powders were far below the statutory criteria, therefore meeting the certified standards for recycling hazardous industrial waste. These results suggest the recycled LCD glass, as a general industrial waste, could properly be used in concrete production.

3.2 Fresh properties

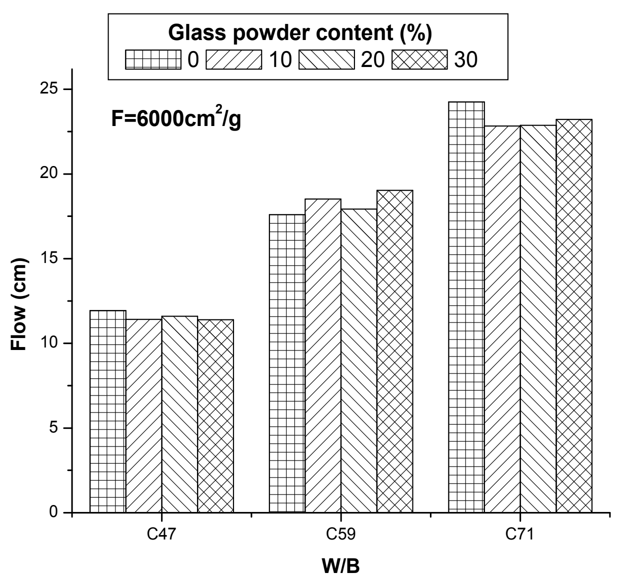


Fig. 1. Relationship between the flow and W/B ratio of waste LCD glass powder cement mortars with the powder fineness = 6000 cm^2/g.

Figure 1 shows the fluidity versus W/B ratio of each group of the mortar specimens. As expected, the mortar fluidity increases with respect to a higher W/B ratio, suggesting that the amount of glass powder has no significant effect to the mortar fluidity. This is primarily caused by the blunt reactivity of glass powders to the hydration of mortar mixtures, even with the powder fineness of 6000 cm^2/g.

Figure 2 shows the relationship between glass powder content and final setting time of the mortar mixtures. As illustrated, the setting time was about 210 to 395 minutes (with glass powder fineness of 6000 cm^2/g). Higher W/B ratio would result in a longer setting time, which was caused by the delay in the hydration process when glass powders were added. It was also observed that the mortar setting time with W/B ratios of 0.59 and 0.71 increased with respect to higher glass powder content. This result suggested that the low water absorption capability of glass powders may have influenced the hydration process of cement mortars. In particular, when a sufficient moisture condition was present (W/B ≥ 0.59), the setting time was significantly extended.

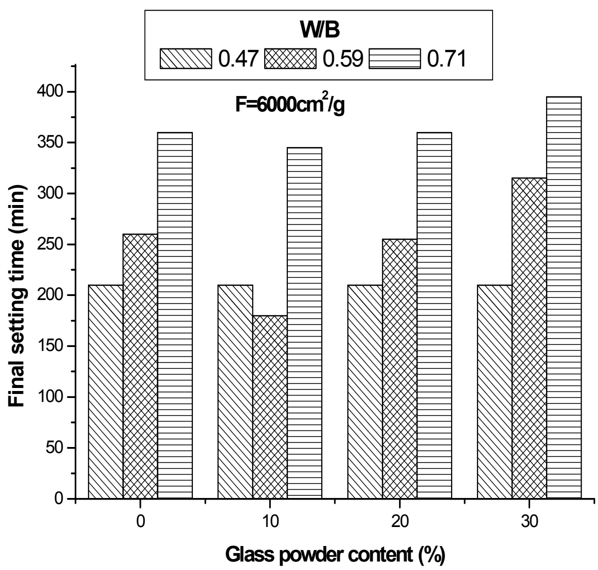


Fig. 2. Relationship between the glass powder content and final setting time of waste LCD glass powder cement mortars with the powder fineness = 6000 cm^2/g.

3.3 Compressive strength

Figure 3 shows the growth of compressive strength of the mortar specimens. There were three plots in the figure, with each plot showing different fineness grade of glass powder inclusion – 1500, 4500, and 6000 cm^2/g, respectively. Cement mortars with different glass powder contents (10%, 20%, and 30%) and identical powder fineness were compared with the control specimen (plain cement mortar), as shown in each plot. All the mortar specimens had a consistent W/B ratio of 0.47. For the mortars with F = 1500 and 4500 am^2/g, specimens showed a lower compressive strength as compared to the control specimen at early age (7 days), while the difference was not significant. The mortar strengths with glass powder fineness = 1500 cm^2/g were shown to closely approach the control group at the age of 91 days, as shown in the left plot. For the group with powder fineness = 4500 cm^2/g

(middle plot), the mortar strength started to surpass the control group after 28 days. In particular, the cement mortar with 10% glass powder replacement has exhibited a compressive strength as high as 63MPa at 91 days. For the mortars with powder fineness = 6000 cm^2/g (right plot), all the testing groups have shown higher compressive strengths than plain cement mortars. Based on the data presented, it is concluded that the inclusion of finer glass powder could significantly enhance the compressive strength of cement mortars, while the optimal powder content is suggested as 10%.

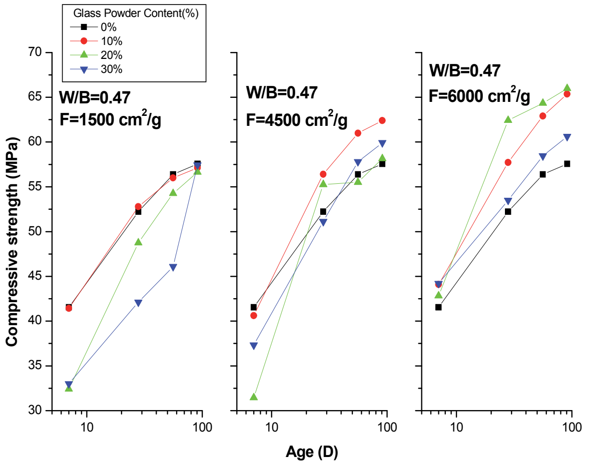


Fig. 3. The compressive strengths of waste LCD glass powder cement mortars (W/B = 0.47) at room temperature.

3.4 Flexural strength

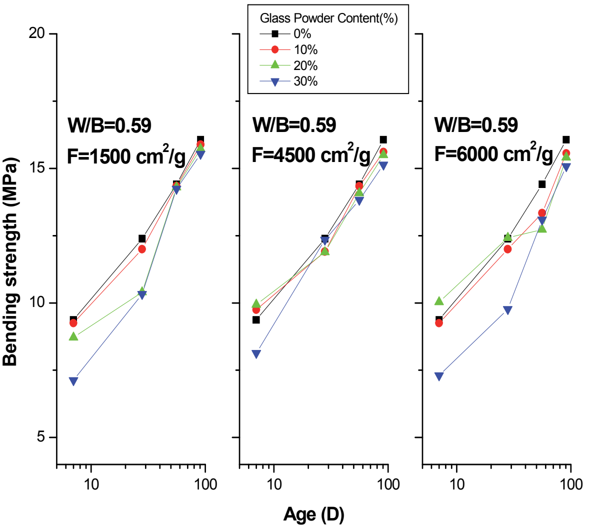


Fig. 4. The flexural strengths of waste LCD glass powder cement mortars (W/B = 0.59) at room temperature.

Similarly, Figure 4 shows the growth of flexural strength of cement mortar specimens with three grades of glass powder fineness (1500, 4500, and 6000 cm²/g) and glass powder content (0%, 10%, 20%, and 30%) under a consistent W/B ratio of 0.59. As seen, the inclusion of glass powder would slightly lower the mortar flexural strength; however, the effect was nearly nullified as the curing age increases. The amount of flexural strength drop depends on the amount of glass powder added (i.e. cement replaced), while the maximum strength drop was shown to be less than 8% even at the age of 91 days.

3.5 Anti-sulfate attack

Figure 5 compares the corrosive weight loss of various glass powder cement mortars with the W/B ratio fixed at 0.59. A complete corrosion cycle has been previously described in section 2.3. Each mortar specimen has experienced five cycles of test. The measurements were taken after each cycle was completed. As expected, the rate of weight loss increased with the cycle of corrosion. Among them, the 20% glass powder group exhibited the most durability (least weight loss) as compared to the other groups. In particular when the powder fineness is 6000 cm²/g, as illustrated in the right plot of the figure, the long-term durability was shown to be even more promising than plain cement mortars.

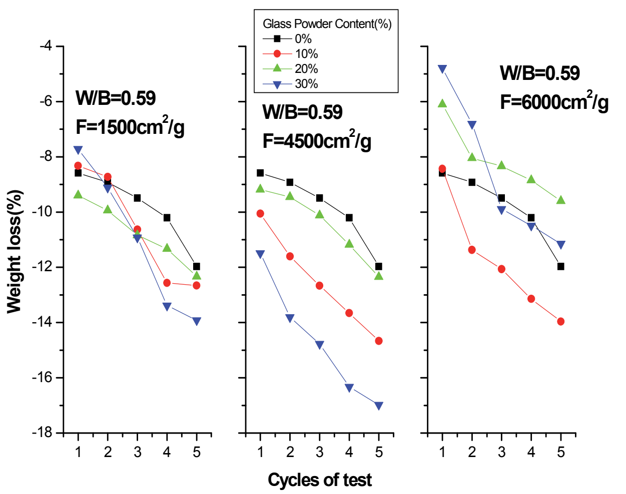


Fig. 5. Weight loss of the sodium sulfate corrosion tests of waste LCD glass powder cement mortars.

3.6 Compressive strength after elevated temperatures

Figure 6-8 show the results of elevated temperature resisting capacity of glass powder cement mortars. Here the powder content was fixed at 20% for all the tests. Although the W/B ratio was shown to have a negative influence to the mortar strength, the inclusion of glass powder appeared to provide a compensating effect to it. As shown in the middle and right plot of Figure 6, the 91 day compressive strengths of the mortars with W/B = 0.59 (F = 6000 cm²/g) and W/B = 0.71 (F = 6000 cm²/g) were 1.8% and 15% higher as compared to their corresponding control groups. Similar to the results discussed in section 3.3, finer glass particles would tend to enhance the compressive strength after experiencing an elevated temperature of 105°C. When the thermal treatment was increased to 580°C, as shown in

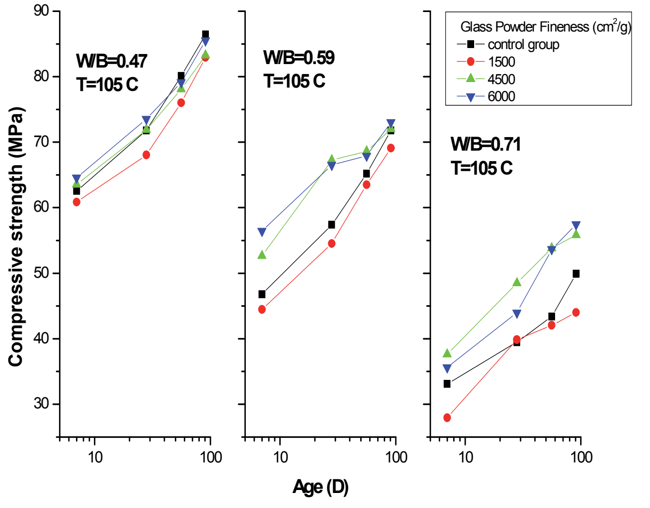


Fig. 6. The compressive strengths of waste LCD glass powder cement mortars after an elevated temperature treatment of 105˚C.

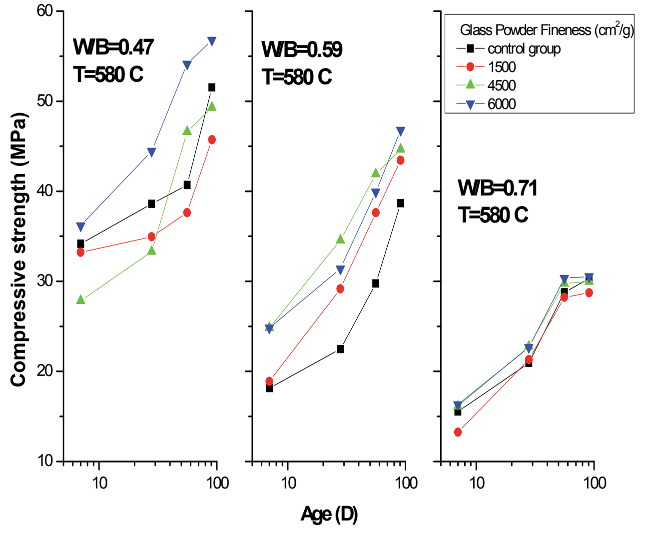


Fig. 7. The compressive strengths of waste LCD glass powder cement mortars after an elevated temperature treatment of 580˚C.

Figure 7, the 91 days compressive strengths of the mortars with W/B = 0.47 (F = 6000 cm^2/g) and W/B = 0.59 (F = 6000 cm^2/g) were 10% and 21% higher as compared to their control groups, respectively. The mortar specimens with the highest W/B ratio of 0.71, on the contrary, exhibited no significant strength enhancement, while the one with F = 6000cm^2/g still behaved a slightly higher strength than others. Figure 8 shows the results when the temperature treatment was further increased to 800˚C. As seen, the effects of W/B ratio and fineness grade to the compressive strength were similar to the case of 580˚C. When the W/B ratio is as high as 0.71, however, the inclusion of glass powder appeared of no

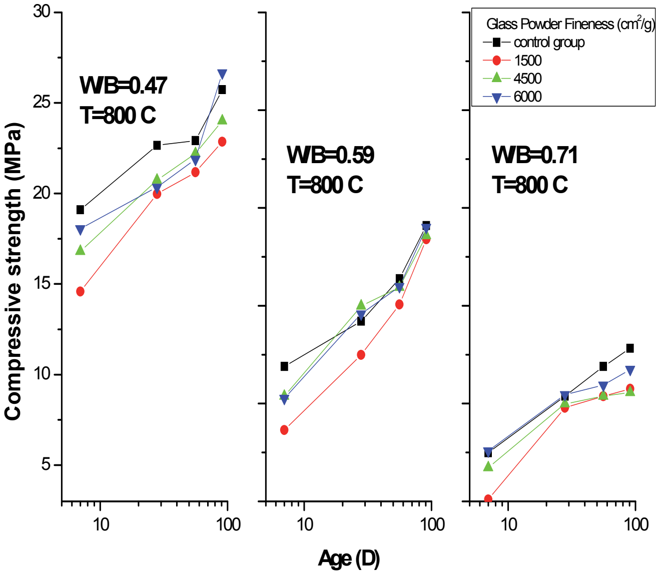


Fig. 8. The compressive strengths of waste LCD glass powder cement mortars after an elevated temperature treatment of 800˚C.

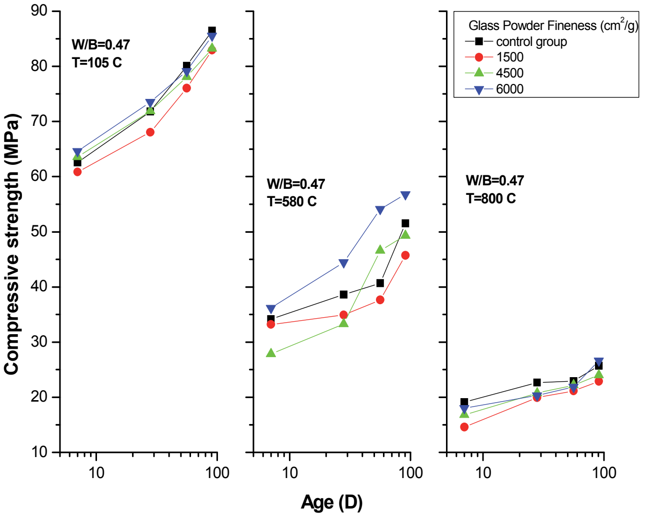
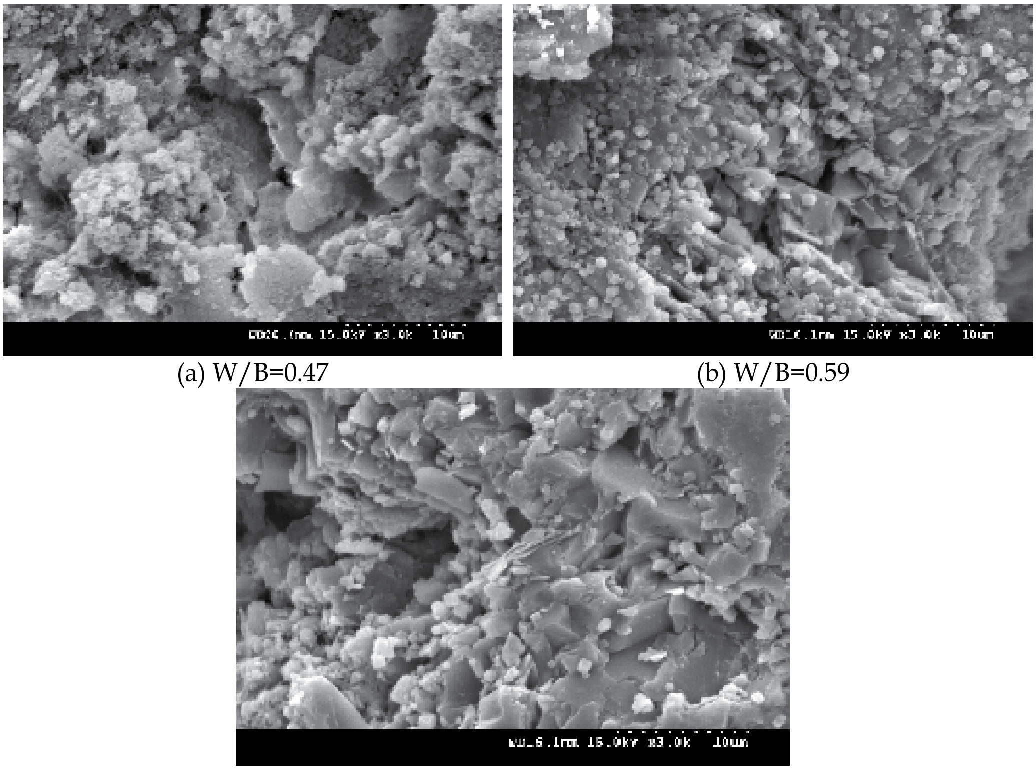


Fig. 9. The compressive strengths of waste LCD glass powder on cement mortars after elevated temperature treatments.

benefit to the mortar strength. Figure 9 summarized the effects of elevated temperatures to the compressive strengths of glass powder cement mortars. It is fairly obvious that higher temperature treatment would result in a lower strength development of cement mortars. This phenomenon was attributed to the cracks and pink spots generated on the surface of the mortar specimens under higher temperature, which would greatly reduce the compressive strength. The 91 days compressive strengths at 105˚C, 580˚C and 800˚C, were

86.44, 51.53 and 25.73MPa, respectively. Although Figure 9 merely summarized the mortars with W/B = 0.47, the other two cases should exhibit similar results.

3.7 Interfacial microstructure (SEM)

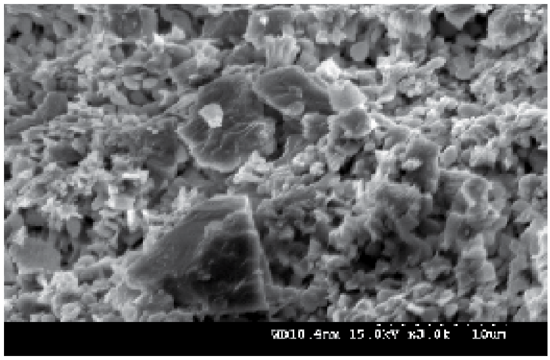


(a) W/B=0.47 (b) W/B=0.59

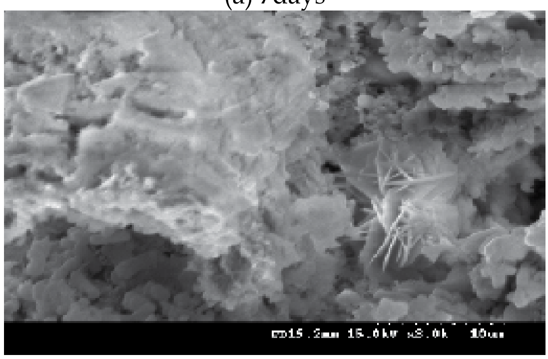
(c) W/B=0.71

Fig. 10. SEM crystal structure diagrams of waste LCD glass powder cement mortars with various W/B ratios, powder fineness = 6000cm²/g, cured at room temperature for 28 days, and thermal treated at105˚C.

Figure 10 shows the SEM images (×3000) of the cement mortar with the glass powder fineness = 6000 cm²/g. The specimens were cured for 28 days and treated with an elevated temperature of 105˚C before SEM observation was performed. As seen, the primary hydration products of the mortar specimens were C-S-H gel and CH hydrolyzed spiked balls. If the temperature continued to increase, water exiting in the C-S-H gel would be forced to release from the pore structures. When the temperature was raised to 800˚C, as shown in Figure 11, the porous structures were mostly granulated and the porosity was significantly enhanced. In particular, a complete hydration was shown to be achieved after 7 days of curing followed by an elevated temperature treatment of 800˚C. It was observed that there still exists certain amount of calcium sulphoaluminate hydrate even when the mortar specimen was cured for 28 days accompanied with a 800˚C thermal treatment. A granulated coarse surface also generated on the CH appearance due to water volatilization at high temperatures.



(a) 7days



(b) 28days

Fig. 11. SEM crystal structure diagrams of waste LCD glass cement mortars with W/B ratio = 0.71, powder fineness = 6000cm^2/g, and treated with an elevated temperature of 800˚C.

4. Conclusions

1. The TCLP test result shows that waste LCD glass powder fairly meets the certification criteria of hazardous industrial waste, suggesting that it could be properly recycled and used for concrete production. For the fresh property examination of the mortars, glass powder content shows no effect to the mortar fluidity, while would increase the final setting time when the W/B ratio exceeds 0.59.
2. Experimental results indicate that substituting 10% of cement by glass powder would gain a very promising compressive strength of the mortars, particularly when the added glass has a powder fineness ≥ 4500 cm^2/g. In real practices, this amount of glass powder substituent could be suggestively used to replace cement.
3. Although the W/B ratio had a negative influence to the mortar strength, the inclusion of glass powder appeared to provide a compensating effect to it. This effect is more prominent when the mortars experienced an elevated temperature. After a thermal treatment of 105˚C, the compressive strengths with W/B = 0.59 (F = 6000 cm^2/g) and W/B = 0.71 (F = 6000 cm^2/g) were 1.8% and 15% higher as compared to their corresponding control groups (plain cement mortars). When the temperature is further raised to 580˚C, the compressive strengths with W/B = 0.47 (F = 6000 cm^2/g) and W/B = 0.59 (F = 6000 cm^2/g) were then 10% and 21% higher as compared to their control groups, respectively.

4. When cured at room temperature for 91 days and treated with an extreme temperature of 800˚C, the group with 20% glass powder content, fineness grade = 6000 cm^2/g, and W/B = 0.47 exhibited the highest compressive strength, exceeding the control group by 3.6%. It should be mentioned that under this circumstance (91 days of curing, 800˚C thermal treatment), the other groups have shown slight to fair decreases in compressive strength.
5. The sulfate corrosion test results indicate that cement mortars with W/B ratio = 0.59 and glass powder content = 20% would behave the best durability performance. In particular when the powder fineness is as fine as 6000 cm^2/g, the long-term durability was shown to be even better than the plain cement mortar.
6. The microstructure observation indicates that the cement mortars would achieve a complete hydration after 7 days of curing and then treated with an elevated temperature of 800˚C. The corresponding SEM image shows that under this circumstance, the porous structures were mostly granulated and the porosity was significantly enhanced.

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Cost-Benefit Analysis of the Clean-Up of Hazardous Waste Sites

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1. Introduction

Hazardous waste, defined as any material that poses a substantial threat to human health, can potentially contaminate all the environmental media: atmosphere, groundwater, surface waters and soil, and through these media can be harmful or even fatal for human health. The prolonged exposure to toxic pollutants such as benzene derivatives, dioxins and trichlorophenol has been associated with acute health effects such as narcosis, skin irritation, or respiratory diseases such as asthma and allergies. Hazardous waste exposure has also been associated with chronic health effects such as leukaemia, liver tumour, lymphomas and, in the case of methylene chloride, premature mortality.

Since the case of Love Canal, New York State, in 1980 an increasing number of cases of hazardous waste mismanagement have been reported. Studies suggest that children are the most vulnerable victims of toxic pollutants. Exposure to compounds increases the likelihood of miscarriage and birth defects. In the Love Canal, for instance, birth defects were found to be twice as likely to occur among those living near the dump site (Goldman et al.1985). In Canada, a large study conducted by Goldenberg *et al.* (1999), suggested that individuals living close to landfill sites have an increased risk of liver, kidney, pancreas cancers and non-Hodgkin's lymphomas.

Another study conducted by Pukkala (2001) in Finland found that the prevalence of asthma was significantly higher in individuals living near landfill sites.

Lack of resources requires policy makers to prioritise competing alternatives. Despite the potential gains for both environmental and human health, it remains uncertain whether the benefits of interventions to clean-up hazardous sites would outweigh the costs. The analytical tool of cost-benefit analysis provides a powerful and transparent method to evaluate and select risk management strategies. Nevertheless, cost-benefit analysis has rarely been used to assess hazardous waste site cleanup interventions. There are several reasons for this: the effects of hazardous waste exposure are often ignored; there are difficulties indentifying the causal link between waste exposure and health effects; and estimating the value of the potential impacts resulting from cleanup interventions. Costs of cleanup interventions are also subject to great uncertainty because it is difficult to quantify them *a priori*, especially where more than one media has been affected by hazardous pollutants. The aim of this chapter is to provide an overview of the major steps necessary to conduct a cost-benefit analysis of cleanup interventions.

2. Economic evaluations of benefit and cost of hazardous site cleanup

Cost-benefit analysis evaluates the social gain associated with a given intervention by comparing the benefits (any increase in welfare) and the costs (any decrease in human well being). The aim of cost-benefit analysis is to maximize the net social benefits:

Max B(Q)-C(Q)

Cost benefit (CB) analysis is used in environmental regulation to determine acceptable levels of risk. Acceptable risk denotes a level that maximizes the difference between total social cost and total social benefits, or in other words, where the marginal social benefits associated with the risk reduction are equal to the marginal social costs of pollution abatement.

In the case of the cleanup of hazardous waste sites, cost benefit analysis is used both to distinguish between interventions offering higher net benefit (difference between cost and benefits) and to identify priority sites for intervention, as in the case of the US Superfund.

CB analysis involves six steps: quantifying the health outcomes associated with waste exposure before and after regulation (hazardous waste site cleanup); assigning monetary values to the number of cases potentially averted by regulation; quantifying the cost of regulation; accounting for the timing of costs and benefits; and comparing the resulting estimates. The final step of CB analysis is to perform sensitivity analysis to evaluate the effect of parameter uncertainty on the study results.

2.1 Health benefits analysis

Several types of benefits result from hazardous waste cleanup. These are: direct benefits, for example reduction in the number of health effects (e.g. asthma cases, lung cancer, malformations); aesthetic benefits, such as decreases in odour; and indirect benefits, such as productivity increase of real estate properties. This chapter focuses on describing how the direct benefits to human health can be quantified using a damage function approach.

As shown in Figure 1 the damage function approach framework uses three types of data: environmental data to identify the potential hazards/pollutants present in the hazardous waste sites; epidemiological data to identify and quantify the health effects associated with the regulatory intervention and economic data to assign a monetary value to negative health outcomes associated to waste exposure.

The first step involves the estimation of the health effects due to pollutant exposure. The second step evaluates the number of health outcomes that can be averted by site cleanup. And the third step multiplies the estimated number of avoidable health outcomes as a result of the regulatory strategy (number of deaths averted per year) by the economic value per health unit (e.g. value of a statistical life).

2.1.1 Quantifying cleanup health benefits

In the majority of cost benefit analyses conducted to evaluate the effects of an environmental regulatory strategy (e.g. air pollution control intervention) the baseline number of health outcomes attributable to pollution exposure is determined using a dose-response function. This function is "an estimate of risk per unit of exposure to pollutant" (EPA, 2010a). The dose-response functions can have different shapes. They can be linear (any change in the pollutant concentration will produce a corresponding change in the health outcome), non-linear (e.g.it can be a sigmoidal curve that starts with an increasing slope but after reaching a maximum value it levels off) and/or can present a threshold dose. For example a study

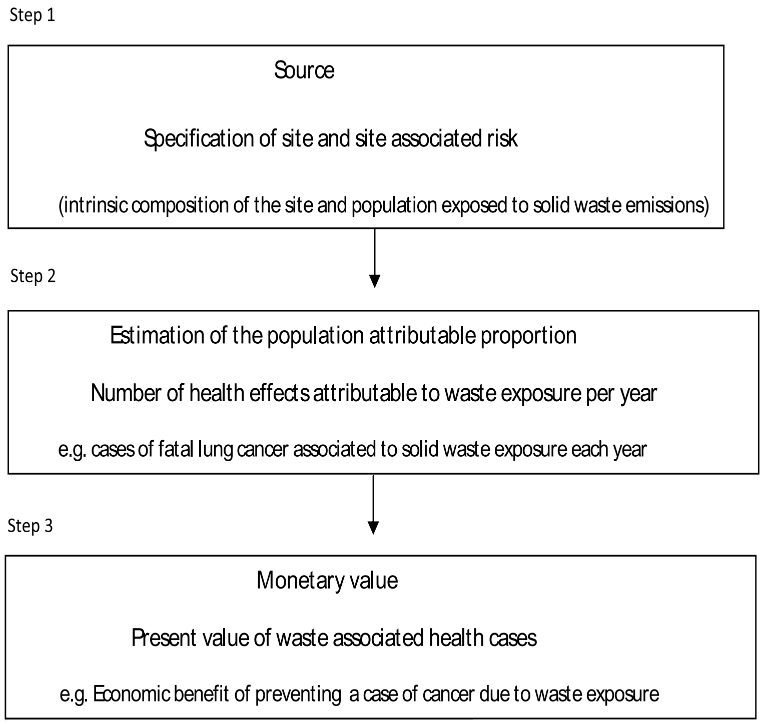

Fig. 1. Damage Function Approach

conducted by Grosse *et al*. (2002) on the relationship between blood lead level and intelligence quotient (IQ) estimates that there is a linear relationship between the blood lead level and the decrease in IQ points (2.57 IQ points for each 10 mg/dL).

Where the effects on health of hazardous waste disposal result from exposure to a single pollutant (e.g. asbestos), the population attributable proportion (PAP), the number of cases that would have not occurred in the absence of pollutant, is estimated using the following formula:

$$\mathrm{PAP} = \left(p - (\mathrm{RR} - 1)\right) / \left(1 + p * (\mathrm{RR} - 1)\right)$$

Where RR is the relative risk of developing the health outcome given pollutant concentration, and p the proportion of the population exposed (e.g. children only).

In the majority of cases, identifying the individual pollutants responsible for the health effects observed in the exposed population is problematic. In the case of landfills or illegal waste disposals, impacts are likely to result from different compounds discharged in the same site. Thus, the PAP is estimated using primary epidemiological data with the following formula:

$$\mathrm{PAP} = \text{Observed number} - \text{Observed number} / \mathrm{SHR}$$

Where SHR is Standardised mortality/hospitalisation ratios (SMR, SHR) that are estimated by dividing the observed cases (e.g. individuals with lung cancer) by the expected cases.

2.1.2 Monetizing health benefits

There are two main methods for placing a monetary value on changes in health: the human capital; and the willingness to pay approach. (Table 1) The human capital approach assumes that the value to society of an individual's life can be measured in terms of future production potential. The willingness to pay (WTP) approach measures how much individuals are willing to pay to decrease the likelihood of a negative outcome.

Basic approach	Main subsets	Evaluation methods
Human capital		Cost of illness
Willingness To Pay	Revealed Preferences	Hedonic wage method Averted expenditures
	Stated Preferences	Contingent Evaluation Stated Choice

Source: Enhealth 2003

Table 1. Methods for valuing health

Based on the human capital approach, the Cost of Illness (COI) method is a measure of the monetary losses due to a negative health outcome (e.g. case of liver cancer). The COI has several advantages. It is straightforward and objective as it both considers all the direct monetary costs of a given health outcome and it does not depend on personal preferences. However, COI tends to underestimate the true value of a health outcome because it does not include the intangible aspects of being ill such as stress, pain and suffering. Additionally, given that the COI values can be estimated only *a posteriori* it is impossible to elicit with this method the values that individuals assign to future environmental risk reductions.

As a result, the most popular approach adopted in cost-benefit analyses is the WTP approach. The WTP method can be divided in two main categories: revealed and stated preferences. The revealed preference method derives values from observed actions of individuals while the stated preference method elicits valuations by asking individuals how much they are willing to pay to reduce the risk of a given health outcome.

Contingent valuation and the Hedonic Wage method have been widely used to estimate the value of saving a statistical life. As can be seen in Table 2 there is great uncertainty regarding the value to adopt for analysis. Estimates vary dramatically among studies and between regions. The meta-analysis by Mrozec and Taylor (2002), for example, suggested a value of a statistical life (VSL) of $2.4 million (in 1998 US$). While in the meta-analysis conducted by Kochi *et al*. (2006) with an empirical Bayes approach the estimated value of a statistical life was $5.4 million.

However, as Pearce (2000) suggests, not all deaths are valued equally and different evaluation techniques can lead to different and often misleading estimates of the VSL.

For example, it has been shown that adopting the human capital approach for assigning a monetary value to mortality risk would underestimate its cost. Although this method is easier to apply as it relies on a simple calculation of visible and easily quantifiable costs it does not consider individual preferences, and willingness to pay for a risk reduction and individual aversion towards death.

Thus, the approach mainly used to estimate the value of a statistical life in environmental health studies has been the willingness to pay approach and in particular, the hedonic wage, and contingent valuation methods.

Estimate	Source	Method	N of studies
\$2.4 million (1998US\$,1990 income)	Mrozec and Taylor (2002)	Hedonic Wage	47
\$5.4 million (2000US\$,1990incomes)	Kochi *et al*. (2006)	Hedonic Wage Contingent Valuation	47 14
\$7.6 million (2002US\$,1990incomes)	Viscusi and Aldy (2003)	CV,HW	33

Table 2. Value of a statistical life

The hedonic wage (HW) method has been widely used in the last decades to estimate the value of a statistical life. The estimation of the WTP (or WTA) in the HW method involves two stages. First, by controlling for productivity and intrinsic quality of the job, the hedonic wage determines the wages associated with the different types of risk according to the equation below:

$$W_k = \alpha + \beta \, \text{Risk}_k + \Sigma \, \lambda_n X_{kn} + \Sigma \, \gamma_m D_m + \varepsilon$$

Where W_k is the wage of the worker k, Risk_k is the risk of death of the worker, n describes human capital and demographic characteristics of the individual X_{kn}, and D_m describes the job characteristics of the individual. The coefficient β (occupational fatal risk) of the risk variable is the additional wage the worker would require to assume an incremental risk of death on the job. Thus according to the hedonic wage method the VSL is estimated as:

$$\text{VSL} = (\delta \text{ w} / \delta \text{r}) * \text{ mean annual wage} * \text{units of fatal risk.}$$

Although this method is widely used in US environmental health studies it presents several disadvantages. The first main disadvantage is that HW does not seem to provide robust and unbiased estimates as it is sensitive to the specification of the wage equation. According to Mrozec and Taylor (2002) studies that control for inter-industry wages have an 85% lower VSL. In addition, it is unclear whether this can be applied only to the occupational risk or whether it can be generalized to the entire spectrum of mortality risks that individuals can face.

Another limitation of the HW method is that it doesn't take account of the characteristics of the person who faces the risk of death nor the risk context. The value assigned to a risk reduction with the HW method is the value for a risk that is immediate, or quite soon in time. While, especially in the context of environmental-related health effects the risk is latent for several years. It is likely that the value that individuals assign to reducing the risk of death in the future is lower than their willingness to pay for a current reduction of risk.

The contingent valuation method on the other hand is a more flexible tool to elicit individuals WTP for fatal risk reduction. According to this method, individuals are asked how much they would be willing to pay for an improvement in their health status or their willingness to accept values for an increased risk. Compared to the COI, this method has the advantage of taking into account the intangible consequences: premature death, the suffering from an illness. In addition, it can be applied also to individuals who are not in the labour force, and can easily account for different types of risk context.

Contextual factors, such as age, health status, income and cultural differences, have been shown to influence how much individuals are willing to pay for a reduction in the risk of an adverse outcome. Several studies demonstrated that older individuals have a decreased willingness to pay for a reduction in mortality risk often referred as the "senior discount" phenomenon. According to Shepard and Zeckauser (1984) the relationship between VSL and age is not linearly decreasing as might be expected but it is an inverted U-shaped relationship which means that the WTP increases until individuals are 40-45 (as their savings increase as well as their income level) and after that peak it decreases with age because the income level decreases and also because their probability of survival declines. Also, the nature of health outcome (death from cancers) and the time of death have been proven to affect individual WTP. Several studies report that individual WTP to avert a case of immediate death (road traffic accident) is lower than for chronic degenerative disease because of the fear and the pain associated with it. As Pearce (2000) suggests, the WTP to avoid cancer is higher than with other types of diseases because of the dread and pain effects associated with this pathology. According to the European Commission (2001) in cases of cancer related mortality VSL should be inflated by 50% to account for the "Cancer premium".

2.2 Cost analysis

Once the potential benefits arising from remediation have been established, it is necessary to quantify the cost of the cleanup, to decide both the stringency of cleanup standards and who should pay for remediation.

It is difficult to evaluate *a priori* the effectiveness of a given remediation strategy and the cessation lag, the time necessary to observe the improvement in health condition of the population exposed (e.g.decrease in the number of malformations).

In general, remediation expenditures can be divided into three main categories: transaction costs borne by agencies (for example EPA in the US superfund) and private parties/polluters (e.g. oil companies); removal actions and long term remediation costs.

Long term remediation cost constitutes the bulk of the overall cost and is highly dependent on the degree of permanence attainable with the cleanup intervention and on the size of the area to reclaim. According to Gupta *et al*.(1998) in the US, it has been estimated that the average cleanup cost is $27 million per site. However, the cost varies according to the type of media that have been contaminated and to the concentration of compounds in the media.

The choice of the technology is also very important. It determines the permanence of the clean-up intervention. In the case of contaminated ground water, the choice of the technology is restricted. The typical method is "to pump and treat" the contaminated water. Following treatment, the water is either released into the aquifer again or released in a river or stream. In the case of contaminated soil remediation there are several alterative options. The first decision is whether to cap the site. Capping soil is the least permanent option (depends on the shelf life of the cap) and has an average cost of $79 per cubic yard (1996 values: Gupta *et al,* 1998). A more permanent option consists of treating the soil *in situ* (costs $231 per cubic yard) (Gupta et al 1998). The third and most permanent option is excavation. In the case of excavation, the removed soil can be transferred to another landfill site or can be further treated and the organic element incinerated. Excavation with offsite treatment is the most expensive option with costs per cubic yard $1,428 (Gupta et al 1998).

2.3 Time adjustment for environmental benefits and costs

The cost and the benefit of a hazardous waste site cleanup, especially in the case of permanent cleanup, materialise over lengthy periods. Thus, discounting plays a crucial role in the estimation of the value of future costs and benefits. Where different types of interventions are compared, discounting future costs and benefits to present values renders them more easily comparable. Discounting implies that the further in the future the benefits and the costs occur, the lower the weight that should be attached to them.
Thus, the general formula of discounting is the following (Pearce *et al.* 2006):

$$W_t = \frac{1}{(1+s)^t}$$

Where w_t is the discount factor for time t and s is the discount rate.
Thus, the conversion of future benefits to a present value can be estimated with the following formula:

$$Present\,Value = \sum Future\,Value_t \times w_t$$

Where economists use discounting to adjust the value of costs and benefits occurring in the future, the standard approach is to assume a constant discount rate common to both costs and benefits. For example, since 1992 the US discount rate suggested as base case for cost-benefit analyses was a fixed at 7% for both cost and benefit estimates. A 3% discount rate was also suggested for sensitivity analysis. The European Commission (2001) recommends for environmental cost benefit analyses the use of a discount rate of 4% and to perform sensitivity analyses using a discount rate of 2 and 4%. However, there has been extensive discussion of whether the discount rate for health benefits should be lower than that applied to monetary costs. Also, where the effects under consideration are long-lived the case for discount rates declining over time has been made.
Mainly due to the lack of empirical studies, there is uncertainty regarding the discount rate to be adopted in the economic evaluation of toxic waste cleanup interventions. A recent study conducted by Alberini *et al*. (2007) in four Italian cities with significant toxic waste problems applied a contingent choice methodology and evaluated that individuals discount future risk with a 7% rate. Recent studies also suggest that the discount rate might not be fixed and that s should be varying with t. According to Viscusi and Hubert (2006) the discount rate shown for improvements in environmental quality does not follow the standard discounted utility model but its pattern is consistent with the hyperbolic model.
Time lag between the cleanup policy and its related benefits is also an important issue. The annual number of health outcomes (for example number of asthma cases) observable in a given area increases after the creation of a waste site which is producing toxic emissions. After a latency period, which denotes the lag between emissions and onset of the negative health effects, the number of health effects will increase at either a proportional or non-proportional rate. Eventually, if both the emission dose and the population exposed remain constant over the years, the incremental number of health outcomes attributable to pollution exposure is likely to remain the same. When a cleanup policy is implemented, there are no immediate reductions in the number of health outcomes. This is referred to as the "cessation lag". Following the cessation lag, there will be a gradual (proportional/non proportional) decline in the effects of the reduced emission on health up to the point where the number of health outcomes is the same as observed before the creation of the waste site.

The formula used to account for both discounting and latency of benefits is the following:

$$\textit{Present value of Benefits} = \lambda * X_a * 1/\left(1+d\right)^l * \left(1-1/\left(1+d\right)^t\right)/d$$

Where: X_a is the number of health endpoints averted by the cleanup, t is the number of years over which the benefits accrue, and d is the discount rate. λ is the WTP for the health outcome a and latency period l, which is the time occurring between the reduction of the exposure and the improvement in the health of the population.

2.4 Cost-benefit evaluation

The main condition for the adoption of a clean-up intervention is that the present value of the benefit exceeds the present value of the cost or that the: Net present value >0. The Net present value (NPV) rule is usually adopted to decide whether to accept or reject an option, to rank different projects and to choose between mutually exclusive projects. An equivalent feasibility test is the benefit cost ratio (BCR) test (Pearce *et al.* 2006):

$$\mathrm{PVB}/\mathrm{PVC} > 1.$$

However, there are differences between the two tests. The first evaluates the excess in benefits and is a more direct way of measuring the social benefits of a cleanup intervention. The second evaluates the benefits per dollar of cost incurred. For example, a cost ratio of 2.2 means that for each dollar invested $2.20 of social benefit is realized (Pearce *et al.* 2006). There is general agreement that BCR can be misleading when used outside the rationing context (when only one project should be evaluated: implemented versus rejected).

2.5 Risk and uncertainty

As mentioned in the previous paragraphs, cost and benefits are difficult to ascertain. In this context, it is important to define risk and uncertainty given that these are often used as interchangeable elements in the literature. Risk denotes the possibility of attaching a probability to costs or benefits that are not known with certainty. Uncertainty denotes a case in which the probability distribution is not available, but crude end points like the min and max are known.

If the decision maker is risk neutral, the expected values of benefits and cost are evaluated. In this case, the net present value equation is as follows (Pearce *et al.* 2006):

$$\mathrm{NPV} = (\textstyle\sum_{I} p_i \times B_i) - (\textstyle\sum_{I} p_j \times C_j)$$

Where Pi is the probability that the benefit Bi occurs and pj is the probability that the cost j occurs.

A recent study evaluating the potential benefit of reducing the pollution exposure in the two industrial areas of Gela and Priolo (Sicily) adopted, for the first time, cost benefit acceptability curves to assess uncertainty in benefit/cost estimates. To build cost benefit acceptability curves Guerriero *et al*. (2011) assign to each parameter a probability distribution (e.g. gamma for cost, normal for excess cases). Then, from each distribution they generate 10,000 Monte Carlo simulation samples. Cost benefit acceptability curves are built plotting the proportion of simulations producing a positive net benefit given a range of remediation cost.

4. Conclusion

Hazardous waste sites are a major environmental problem. There is a large body of literature showing an association between hazardous waste (mis)management and negative health outcomes. Substances resulting from industrial production (e.g. arsenic, cadmium and mercury) once released into landfills without proper treatment can be fatal for the populations exposed. In the US, the public has ranked toxic wastes sites as the number one national environmental priority. A recent study of a contaminated site in the Italian region of Campania, found that 87% of survey respondents believed that they are going to suffer from cancer because of waste exposure (Cori & Pellegrino 2011). Responding to public concerns, national reclamation projects have been created in several countries, e.g. Superfund program in the US, and programma nazionale di bonifica in Italy. The objective of these programs is collecting public and private resources to prioritize the clean-up of hazardous waste sites. Cost benefit analysis is a transparent decision informing procedure to prioritize the cleanup of those sites that for a given remediation budget would allow to produce the highest benefit in terms of negative health outcomes averted.
Despite the potential benefits resulting from the application of cost benefit analysis in waste management there are few empirical studies using this tool. The study conducted by Hamilton and Viscusi (1999) evaluating the cost effectiveness of EPA Superfund decisions showed that the majority of clean-up decisions are ineffective and highlights the importance of conducting site level analysis. Further studies conducted in US found that other factors such as media coverage were prevailing in determining the stringency of clean-up standards and the selection of clean-up sites/size. As long as the true benefits and costs of cleanup interventions are ignored resources will be allocated inefficiently. Despite measurement problems and the equity issues, cost-benefit analysis should be conducted routinely to address National Superfund's decisions. (Zimmerman and Rae, 1993).

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Part 4

Software Applications

22

Benefits from GIS Based Modelling for Municipal Solid Waste Management

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1. Introduction

Waste management issues are coming to the forefront of the global environmental agenda at an increasing frequency, as population and consumption growth result in increasing quantities of waste. Moreover, technological development often results in consumer products of complex composition, including hazardous compounds, which pose extra challenges to the waste management systems and environmental protection at the end of their useful life, which may often be fairly short (e.g. cell-phones and electronic gadgets). These end-of-pipe challenges are coupled with the deepening understanding that the Earth's natural resources are finite by nature and their current exploitation rate unsustainable, even within a midterm perspective. The self-cleaning capacity of the Earth systems is often also viewed as a «natural resource» under stress, with climate change being the most pronounced expression of this risk.

In the context of the above mentioned challenge a New Paradigm for waste management has emerged, shifting attention to resources efficiency and minimisation of environmental impacts throughout the life cycle of waste management, from waste prevention to safe disposal. This is best expressed, but not confined, in the relevant EU policy and legislation (e.g. the Thematic Strategy on the prevention and recycling of waste, the Thematic Strategy on the Sustainable Use of Natural Resources and the revised Waste Framework Directive, WFD-2008/98/EC). Especially the latter is of particular interest as it has a legally binding nature for all EU member states and sets a benchmark which is often also taken into consideration by the waste management systems of non-EU countries. The WFD reaffirms the need to move waste management higher in the so called "waste hierarchy", preferring, in this order, prevention, reuse, recycling and energy recovery over disposal. Separate collection for dry recyclables in municipal solid waste (MSW) should be implemented while separate collection of biowaste should be promoted (although no specific legislative requirements are set) (Nash, 2009).

Overall, EU and national waste management policies and legislation in many parts of the world are becoming increasingly demanding for the providers of these services, namely municipalities and their associations, demanding high recovery and recycling rates for a wide range of materials and goods, high diversion targets for the biodegradable fraction of the waste, advanced treatment processes, long after-care periods for existing and future landfills etc (COM, 2005; Lasaridi, 2009). Moreover, this increased level of service will need to be provided at the minimum possible cost, as the public will not be able to bear large increases in its waste charges and municipalities are increasingly being required to benchmark their performance, to ensure they offer their waste management services at the most efficient manner (Eunomia, 2002; Karadimas et al., 2007). The current economic crisis inevitably intensifies this need.

The need for improved performance at low costs is not restricted to developed countries seeking to apply increasingly complex separate waste collection, treatment and recovery systems. Under a different context, it also exerts its pressure to the municipal services of the developing countries, which strive to ensure waste collection and public health protection for the large populations of highly urbanised areas with severe infrastructure and economic limitations (Gautam & Kumar, 2005; Ghose et al., 2006; Kanchanabhan et al., 2010; Vijay et al., 2005).

Local authorities (LAs) constitute worldwide the main providers of municipal solid waste (MSW) management services, either directly or indirectly through subcontracting part or all of these services. Especially waste collection and transport (WC&T) are typically provided at the local municipality level and constitute the main interface between the waste generator and the waste management system. Assessing the different components of the solid waste management costs is a complex, poly-parametric issue, governed by a multitude of geographic, economic, organisational and technology selection factors (Eunomia, 2002; Lasaridi et al., 2006). However, in all cases WC&T costs constitute a significant component of the overall waste management costs, which may approach 100% in cases where waste is simply dumped. For modern waste management systems WC&T costs vary in the range of 50-75% of the total, which overall is significantly higher, as advanced treatment and safe disposal take their own, large share of the total costs (Sonesson, 2000).

Therefore, the sector of WC&T attracts particular interest regarding its potential for service optimisation as (a) waste management systems with more recyclables' streams usually require more transport (Sonesson, 2000) and (b) this sector, even for commingled waste services only, already absorbs a large fraction of the municipal budget available to waste management (Lasaridi et al., 2006). Optimisation of WC&T making use of the novel tools offered by spatial modelling techniques and geographic information systems (GIS) may offer large savings, as it is analysed further in this chapter. In spite of their proved utility and a significant development of the relevant research in the last decades in many parts of the world, including most Greek local authorities, WC&T is typically organised empirically and in some cases irrationally, under public pressures.

The aim of this chapter is to present a methodology for the optimisation of the waste collection and transport system based on GIS technology. The methodology is applied to the Municipality of Nikea (MoN), Athens, Greece based on real field data. The strategy consists of replacing and reallocating the waste collection bins as well as rescheduling the waste collection via GIS routing optimisation. The benefits of the proposed strategy are assessed in terms of minimising collection time, distance travelled and man-effort, and consequently financial and environmental costs of the proposed collection system.

2. The role of GIS for sustainable waste management

Geographic Information Systems (GIS) are one of the most sophisticated modern technologies to capture, store, manipulate, analyse and display spatial data. These data are usually organised into thematic layers in the form of digital maps. The combined use of GIS with advanced related technologies (e.g., Global Positioning System – GPS and Remote

Sensing - RS) assists in the recording of spatial data and the direct use of these data for analysis and cartographic representation. GIS have been successfully used in a wide variety of applications, such as urban utilities planning, transportation, natural resources protection and management, health sciences, forestry, geology, natural disasters prevention and relief, and various aspects of environmental modelling and engineering (among others: Brimicombe, 2003). Among these applications, the study of complex waste management systems, in particular siting waste management and disposal facilities and optimising WC&T, have been a preferential field of GIS applications, from the early onset of the technology (Esmaili, 1972; Ghose et al., 2006; Golden et al., 1983; Karadimas et al., 2007; Sonesson, 2000). Nowadays, integrated GIS technology has been recognised as one of the most promising approaches to automate the process of waste planning and management (Karadimas & Loumos, 2008).

As mentioned above, the most widespread application of GIS supported modelling on waste management lies in the areas of landfill siting and optimisation of waste collection and transport, which are discussed in detail in the following section. Additionally, GIS technology has been successfully used for siting of recycling drop-off centres (Chang & Wei, 2000), optimising waste management in coastal areas (Sarptas et al., 2005), estimating of solid waste generation using local demographic and socioeconomic data (Vijay et al., 2005), and waste generation forecasting at the local level (Dyson & Chang 2005; Katsamaki et al., 1998).

2.1 GIS-based modelling for landfill selection

The primary idea of superimposition of various thematic maps in order to define the most suitable location according to the properties of the complex spatial units derived after the map overlay, was first introduced in the late 60’s (McHarg, 1969). This idea was applied next within the context of early GIS in many optimal siting applications (Dobson, 1979; Kieferand & Robins, 1973). The allocation of a landfill is a difficult task as it requires the integration of various environmental and socioeconomic data and evolves complicated technical and legal parameters. During this process the challenge is to make an environmentally friendly and financially sound selection. For this purpose, in the last few decades, many studies for landfill site evaluation have been carried out using GIS and multicriteria decision analysis (Geneletti, 2010; Higgs, 2006; Nas et al., 2010; Sener et al., 2006), GIS in combination with analytic hierarchy process (Saaty, 1980) – AHP (Vuppala et al., 2006; Wang et al., 2009), GIS and fuzzy systems (Chang et al., 2008; Gemitzi et al., 2007; Lofti et al., 2007), GIS and factor spatial analysis (Biotto et al., 2009; Kao & Lin, 1996), as well as GIS-based integrated methods (Hatzichristos & Giaoutzi 2006; Gómez-Delgado & Tarantola 2006; Kontos et al., 2003, 2005; Zamorano et al., 2008).

A large fraction of these applications produce binary outputs while most recent ones aim at evaluating a ”suitability index” as a tool for ranking of the most suitable areas (Kontos et al., 2005). The main steps of a typical GIS – based landfill allocation model (fig.1) are as following.

a. Conceptualisation of the evaluation criteria and the hierarchy of the landfill allocation problem. This step is dedicated to the selection of the criteria related to the problem under investigation.
b. Creation of the spatial database. Here, the development of GIS layers for the modelling is implemented. These layers correspond to the primary variables.

c. Construction of the criteria – layers within the GIS environment. Criteria maps are primary or secondary variables.
d. Standardisation of the criteria – layers. This step includes reclassification of the layers in order to use a common scale of measurement. Most often, the ordinal scale is used.
e. Estimation of the relative importance for the criteria. This estimation is implemented by weighting, e.g. with the use of Analytic Hierarchy Process (AHP) and pair wise comparison between variables.
f. Calculation of the suitability index. A standard procedure for this step is the weighted overlay of the standardised criteria/layers.
g. Zoning of the area under investigation is the next phase of the modelling. This classification action is based on the suitability index and reveals the most suitable areas for the application.
h. Sensitivity analysis and validation of the model.
i. Final selection – land evaluation.

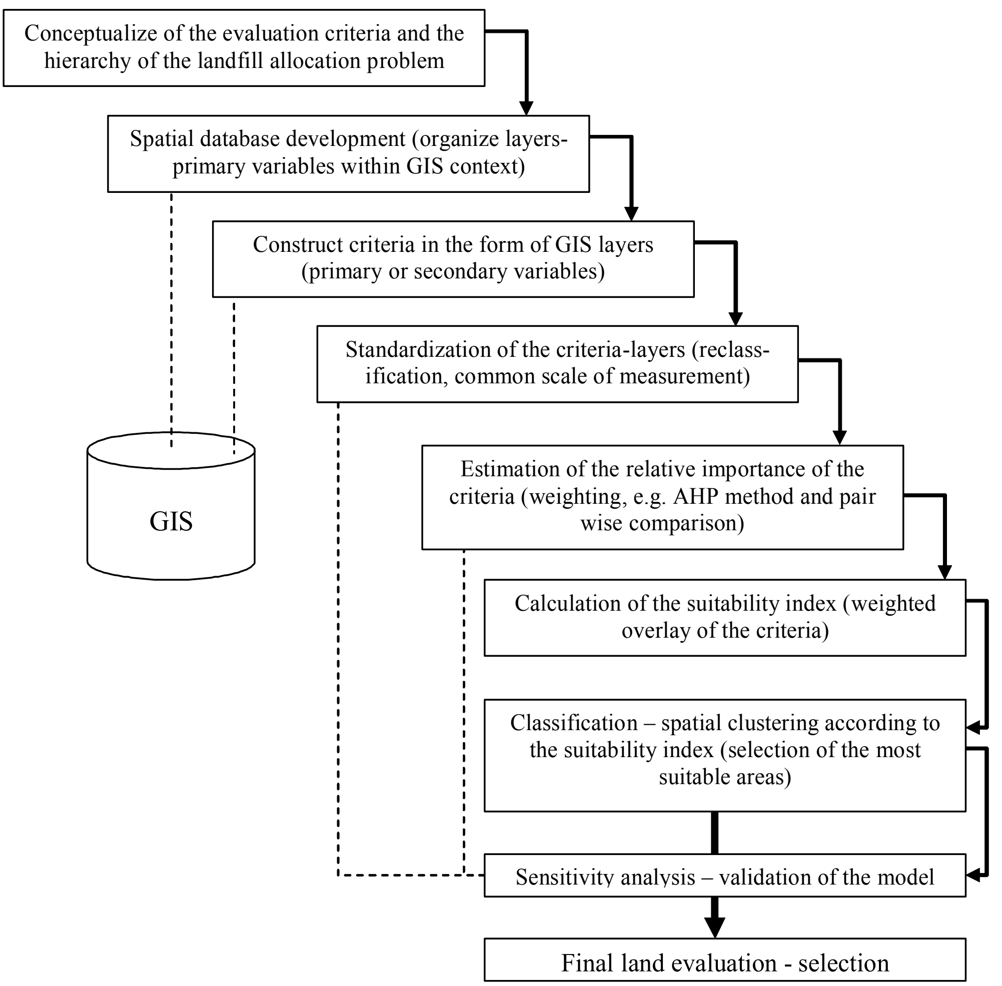

Fig. 1. Landfill site selection. A GIS approach.

It should be noticed that for most of the aforementioned functions the geographic background (in digital format) of the area under investigation is required. Figure 1 demonstrates the data flow of the adopted procedure. Sumanthi et al. (2008) underline that the main advantages of applying GIS technology in the landfill siting process are: "*the selection of objective zone exclusion process according to the set of provided screening criteria, the zoning and buffering function, the potential implementation of 'what if' data analysis and investigating different potential scenarios related to population growth and area development, as well as checking the importance of the various influencing factors etc., the handling and correlating large amounts of complex geographical data, and the advanced visualization of the output results through graphical representation*."

Additionally, the incorporation of various spatial analysis methods, such as geostatistics, analytical hierarchy process, fuzzy logic modelling and many others, constitutes a major advantage of a GIS-based modelling approach. Finally, a particularly useful option of a GIS-based decision making model is the combination of experts knowledge with the opinions of citizens and stakeholders (Geneletti, 2010).

3. GIS modelling for the optimisation of waste collection and transport

The optimisation of the routing system for collection and transport of municipal solid waste is a crucial factor of an environmentally friendly and cost effective solid waste management system. The development of optimal routing scenarios is a very complex task, based on various selection criteria, most of which are spatial in nature. The problem of vehicle routing is a common one: each vehicle must travel in the study area and visit all the waste bins, in a way that minimises the total travel cost: most often defined on the basis of distance or time but also fuel consumption, CO_2 emissions etc. This is very similar to the classic Travelling Salesman Problem (TSP) (Dantzig et al., 1954). However, the problem of optimising routing of solid waste collection networks is an asymmetric TSP (ATSP) due to road network restrictions; therefore adaptations to the classic TSP algorithm are required, making the problem more complex.

As the success of the decision making process depends largely on the quantity and quality of information that is made available to the decision makers, the use of GIS modelling as a support tool has grown in recent years, due to both technology maturation and increase of the quantity and complexity of spatial information handled (Santos et al., 2008). In this context, several authors have investigated route optimisation, regarding both waste collection in urban and rural environments and transport minimisation, through improved siting of transfer stations (Esmaili, 1972), landfills (Despotakis & Economopoulos, 2007) and treatment installations for integrated regional waste management (Adamides et al., 2009; Zsigraiova et al., 2009).

Optimisation of WC&T making use of the novel tools offered by spatial modelling techniques and GIS may provide significant economic and environmental savings through the reduction of travel time, distance, fuel consumption and pollutants emissions (Johansson, 2006; Kim et al., 2006; Sahoo et al., 2005; Tavares et al., 2008). These systems are particularly rare in Greek local authorities, where WC&T is typically organised empirically and in some cases irrationally, under public pressures.

According to Tavares et al. (2008) "*effective decision making in the field of management systems requires the implementation of vehicle routing techniques capable of taking advantage of new technologies such as the geographic information systems*". Using GIS 3D modelling in the island of Santo Antao, Republic of Cape Verde, an area with complex topography, they achieved up to 52% fuel savings compared to the shortest distance, even travelling a 34% longer distance. Nevertheless, most of the previous work relating to optimal routing for solid waste collection is based on the minimisation of the travelled distance and/or time (Apaydin & Gonullu, 2007; Lopez et al., 2008), which is considered a sufficient calculator parameter for fuel consumption and emissions minimisation in flat relief (Brodrick et al., 2002).

Sahoo et al. (2005) presented a comprehensive route-management system, the WasteRoute for the optimal management of nearly 26000 collection and transfer vehicles that collect over 80 million tons of garbage every year for more than 48 states of USA. The Implementation of WasteRoute across the USA from March 2003 to the end of 2003 yielded 984 fewer routes, saving $18 million.

Alvarez et al. (2008) presented a methodology for the design of routes for the "bin to bin" collection of paper and cardboard waste in five shopping areas of the city of Leganés (Community of Madrid, Spain). Their proposed system was based on GIS technology and optimised urban routes according to different restrictions. From the comparison of their system with the previous situation they concluded that the proposed "bin to bin" system improved the quality of the paper and cardboard in the containers, avoiding overflow and reducing the percentage of rejected material.

Teixeira et al. (2004) applied heuristic techniques to solve a collection model in order to define the geographic zones served by the vehicles, as well as the collection routes for recyclable waste collection of the centre-littoral region of Portugal. The study indicated that proper modelling of the collection procedure can provide cost effective solutions.

Nuortio el al. (2006) developed a GIS-based method for the optimisation of waste collection routes in Eastern Finland. They estimated an average route improvement in comparison with the existing practice of about 12%. Moreover they proposed a combination of routing and rescheduling optimisation. This combination in some cases introduced extremely significant savings (~40%). They concluded that by allowing rescheduling it is possible to significantly increase the improvement rate.

Karadimas & Loumos (2008) proposed a method for the estimation of municipal solid waste generation, optimal waste collection and calculation of the optimal number of waste bins and their allocation. This method uses a spatial Geodatabase, integrated in a GIS environment and was tested in a part of the municipality of Athens, Greece. After the reallocation of the waste bins, their total number was reduced by more than 30%. This reduction had a direct positive impact on collection time and distance.

Chalkias & Lasaridi (2009) developed a model in ArcGIS Network Analyst in order to improve the efficiency of waste collection and transport in the Municipality of Nikea, Athens, Greece, via the reallocation of waste collection bins and the optimisation of vehicle routing in terms of distance and time travelled. First results demonstrated that all the examined scenarios provided savings compared to the existing empirical collection organisation, in terms of both collection time (savings of 3.0% -17.0%) and travel distance (savings of 5.5% - 12.5%).

Apaydin & Gonullu (2006) developed an integrated system with the combination of GIS and GPS technology in order to optimise the routing of MSW collection in Trabzon city, northeast Turkey. The comparison of the proposed optimised routes with the existing ones revealed savings of 4–59% in terms of distance and 14-65% in terms of time, with a benefit of 24% in total cost.

Finally, Kanchanabhan et al. (2008) attempted to design and develop an appropriate storage, collection and routing system for Tambaram Municipality in South Chennai, India using GIS. The optimal routing was investigated, based on population density, waste generation capacity, road network, storage bins and collection vehicles. They roughly estimated 30% cost-savings with this approach.

4. The Nikea case study, in Greece

The total cost for waste collection and transport (WC&T) in Greece frequently accounts for more than 70% of the total municipal solid waste (MSW) management costs. Thus, it is crucial to improve the WC&T system through routing optimisation.

Here we present a general methodology for the optimisation of the waste collection and transport system, based on GIS, technology for the municipality of Nikea (MoN), Athens, Greece. This methodology was developed using standard GIS and network analysis procedures in order to improve the efficiency of WC&T in the study area via: (a) the reallocation of waste collection bins; and (b) the optimisation of vehicle routing in terms of distance and time travelled, via GIS routing. The outputs of various different scenarios examined are finally compared with the empirical routing, which is the current vehicle routing practice. Benefits are assessed in terms of minimising collection time, distance travelled and man-effort, and, consequently, financial and environmental costs of the collection system.

In Greece Local Authorities (LAs) are by law responsible for waste management (Decrees 25/1975 and 429/1976). Waste collection and transport are provided at the individual municipality level, usually directly through their Waste Management Department. Currently, WC&T of commingled MSW in the country is responsible for a large portion of the total waste management cost (70% - 100%), which is considerably higher than the typical values, of between 50 and 75%, reported for modern waste management systems (Sonesson, 2000). This is observed because the largest fraction of the waste stream is currently landfilled at very low cost, without pre-treatment for materials and/or energy recovery, while in some cases illegal dumping may be still practiced (Lasaridi, 2009).

4.1 The study area and the existing collection system

The MoN (Fig. 2) is one of the largest in the Attica Region, lying in the SW part of Athens metropolitan area. It has a permanent population of 95,798 habitants according to the 2001 Census (National Statistical Service of Greece - NSSG, 2001) and a total area of 6.65 km^2. Nikea is a typical Greek urban municipality, characterised by multi-storey apartment buildings, combined by lower multiple dwellings (2-4 apartments) and mixed residential and commercial land uses in many neighbourhoods. The annual MSW production in MoN is estimated at 45,625 tn, or 1.30 kg/ca/d.

Waste collection is carried out mechanically, using 12,107 wheelie bins and 17 rear-end loaded compaction trucks with 9 tn average capacity. Most of the bins are small, of 120 and 240 L capacity, but a few larger ones exist in some central points. The total storage capacity of the bin system is 3.4 million litres. The crew size on the collection vehicle is three persons, a driver who never leaves the truck (as required by safety regulations) and two workers who move and align the bins with the hydraulic lifting mechanism of the truck.

Nevertheless, due to traffic restrictions and narrow roads, it is estimated that only 70% of the bins are really mechanically collected, with the content of the rest being manually transferred in other bins, by an extra worker walking ahead of the collection vehicle. The Municipality is empirically divided into 15 sectors (collection zones), each of which is further divided into two sub-sectors. Waste is collected in each sub-sector four times per week.

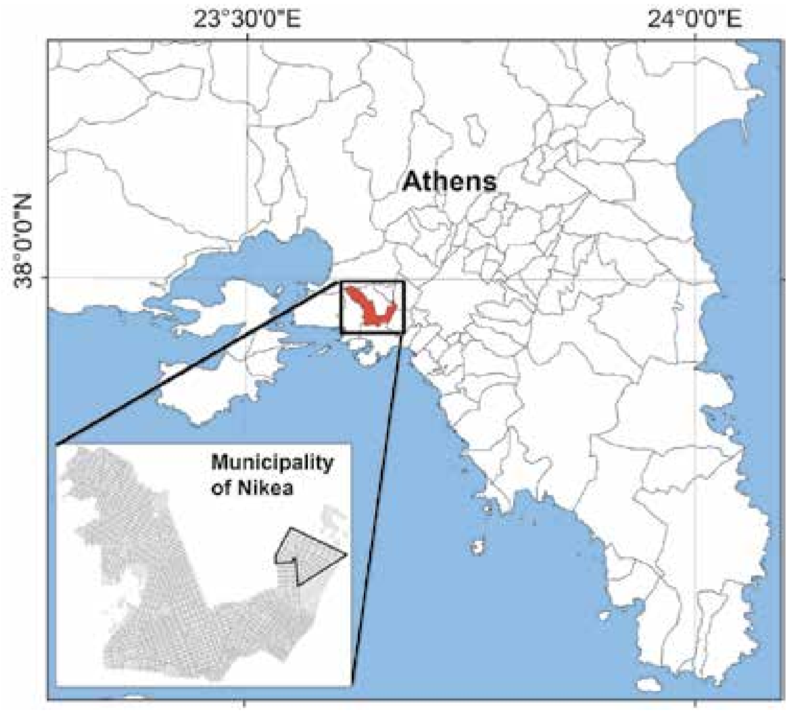

Fig. 2. The study area: Municipality of Nikea, Athens, Greece.

This work applies the developed waste collection and transport optimisation methodology in a typical sector (Sector 1) of the municipality with mainly residential land uses. However, some commercial establishments, schools, stadiums and parks are also found in the area. The served equivalent population in Sector 1 (i.e. taking into account the MSW load created by non-residential land uses) is 6,790 people, divided in 63 parcels (building blocks). The total average waste production is 2,610 ton/yr, according to the weighing sheets of the collection vehicles in the period 2005-2007. This corresponds to an average daily commingled waste production of 1.053 kg/ca eq. This is not in contrast with the municipality average reported above, as the former is calculated on the basis of the 2001 census population, while the latter also takes into account the equivalent population corresponding to the non-residential land uses.

In the current waste collection system, 714 bins are located in Sector 1 (Fig.3), of which 501 are mechanically collected, with total capacity of 157,000 L. The content of the rest is manually transferred to the mechanically collected ones by the extra worker mentioned above. Since Sector 1 is rather flat (mean elevation ~ 50 m) it is assumed that fuel consumption and emissions are linearly related to collection time (Brodrick et al., 2002).

For waste collection purposes Sector 1 is divided into two sub-sectors both served by one waste collection vehicle. Waste in each sub-sector is collected four times per week, in alternate week days, resulting into eight collection trips per week. Collected waste is disposed of at the Fyli landfill site, about 25 km north-west from Sector 1. The key points to the proposed optimisation approach are: a) the replacement of the existing large number of small bins (120 and 240 L) with a reduced number of larger bins (1100 L); b) the resectorisation; and finally, c) the optimal routing. Using the collected data and the analytical tools of the GIS software, specific proposals are developed regarding the optimisation of the existing WC&T system of commingled MSW. For results assessment both the vehicle trip within the sector and travel to and from the landfill are considered.

Fig. 3. Waste bins in the study area.

4.2 Data collection and spatial database description

To efficiently manage the municipal solid waste system, detailed spatial information is required. This information is related to the geographical background of the area under investigation, as well as to spatial data related to the waste collection procedure. A large amount of waste management data for the period 1998-2007 has been collected and statistically analysed regarding the static and dynamic data of each existing collection program: population density; waste generation rate for mixed waste and for specific waste streams; number, type and positions of waste bins; the road network and the related traffic; the current routing system of the collection vehicles; truck capacities and their characteristics; and the geographic boarders and characteristics of the waste collection sectors. The range of data acquired and utilised is illustrated in Table1.

For the optimisation of the collection process a spatial geodatabase was constructed, in a standard commercial GIS environment (ArcGIS, ESRI). This choice ensures compatibility with the available data from the municipality and access to many network analysis routines available from the software. The content of the spatial database is summarised in Table 2.

Background spatial data for road network, existing routes, bins and building parcels were obtained from MoN. These data were updated with field work and other non spatial data such as road name, road type, vehicle average speed, travel time, road slope, bin number, bin type/capacity, bin collection time were added. Furthermore, special attributes of road network were registered. These attributes include traffic rules, traffic marks, topological conditions and special restrictions (e.g. turn restrictions) in order to efficiently model the real world road network conditions.

Data	*Source*
Study area boundary	(MoN Corporation)
Detailed urban plan of the municipality	(official toposheet plan)
Population density distribution	(National Statistical Service of Greece: NSSG)
Land use of the study area	(NSSG)
Satellite image of the municipality	(Google Earth)
Road network of the study area	(official toposheet plan, , field work)
Road class information: restrictions and traffic volume details	(official toposheet plan, MoN Corporation, field work)
Location of waste bins	(MoN Corporation, field work)
Capacities of bins	(MoN Corporation, field work)
Time schedule for the collection process	(MoN Corporation, field work)
Existing collection routes	(MoN Corporation, field work)
Vehicle speed, fuel consumption, CO_2 and other gas emissions of the compactors	(MoN Corporation, field work, literature).

Table 1. Data collected and their source.

Spatial Data	*Type*	*Geometry*
Road network	vector	Line
Waste bins	vector	Point
Urban plan / parcels	vector	Polygon
Existing run routes	vector	Line
Street address	tabular	-
Road network attributes / restrictions	tabular	-
Waste bins' attributes	tabular	-
Population data	tabular	(join with parcels)
Land use data	tabular	-
Satellite image of the MoN	Raster	-

Table 2. The spatial database - type of data and corresponding geometry.

4.3 Methodology

The key point of the proposed analysis is GIS technology. GIS provides a powerful context to import, manage and analyse spatially based data. The methodology implemented in this study comprised of three general steps (Fig. 4). Step 1 establishes the spatial database of the study area as described previously. Step 2 is dedicated on the reallocation of waste collection bins with the use of GIS spatial analysis functions. Finally, Step 3 consists of the

waste collection routing optimisation for minimum time, distance, fuel consumption and gas emissions. The waste collection optimisation model was developed with the use of ArcGIS 9.2 Network Analyst (NA) GIS software.
To analyse the spatial data for the optimisation of the waste collection scheme in MoN, a spatial database (SDB), within a GIS framework, was constructed, as previously described, using: (a) analogue maps from MoN; (b) digital data from various official providers (e.g. National Statistical Service); (c) data derived from field work /on-site data capture with the use of GPS technology.

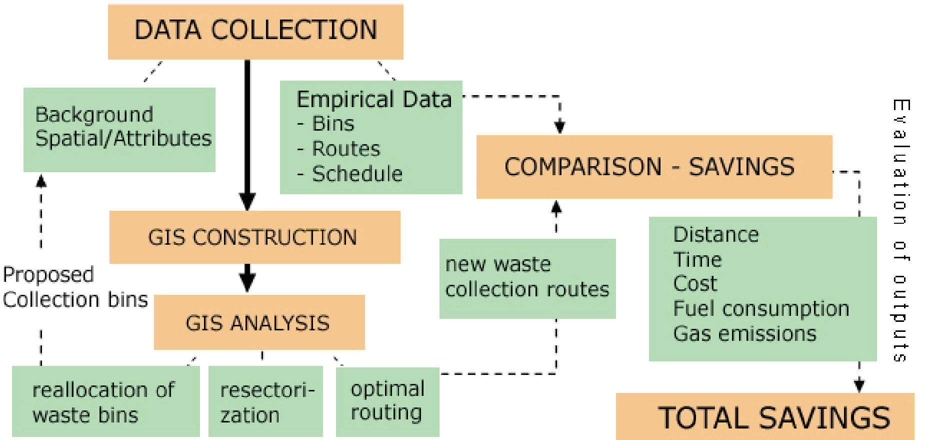

Fig. 4. Data flow of the proposed methodology

4.3.1 Reallocation of waste collection bins and resectorisation

The next phase of the proposed methodology is related to the reallocation of waste collection bins. This analysis was implemented in a GIS environment with the use of the proper spatial analysis functions. The allocation of waste collection bins in their newly proposed locations was based on the following criteria /restrictions:

i. On the basis of the population density and the type of buildings in the study area, bins of 1100L capacity were considered preferable, in order to minimise the number of required bins and vehicle stops. This is the typical bin type used in most Municipalities in the wider Athens area.
ii. The required number of bins (N) was calculated to cover the waste production of the sector for a five trips per week schedule (D=7/5), assuming a waste density in the bin of ρ=110 kg.m^{-3}, and a coefficient of filling the bin, ε = 0.80 of its capacity, according to the equation (1):

$$N = W_D x D / (V x \rho x \varepsilon) \tag{1}$$

where WD (kg) is the daily waste quantity and V (m^3) is the bin capacity. A 10% safety margin was added to this number (Panagiotakopoulos, 2002).
Thus, instead of the existing 501 bins of various sizes (§2.2) Sector 1 is covered by 142 large bins (1100L).
iii. Next, these bins are allocated in the study area according to the following rules: a) allocate bins on the road network (intersections are preferable); b) install proposed bins

near an existing bin location (in a buffer zone of 60 m radius); and, c) allow the placement of more than one bin in the same intersection. The number of bins sharing the same intersection point is related to the land use and population of the covered area.

Figure 5 illustrates the proposed reallocation of waste bins in the sector under investigation.

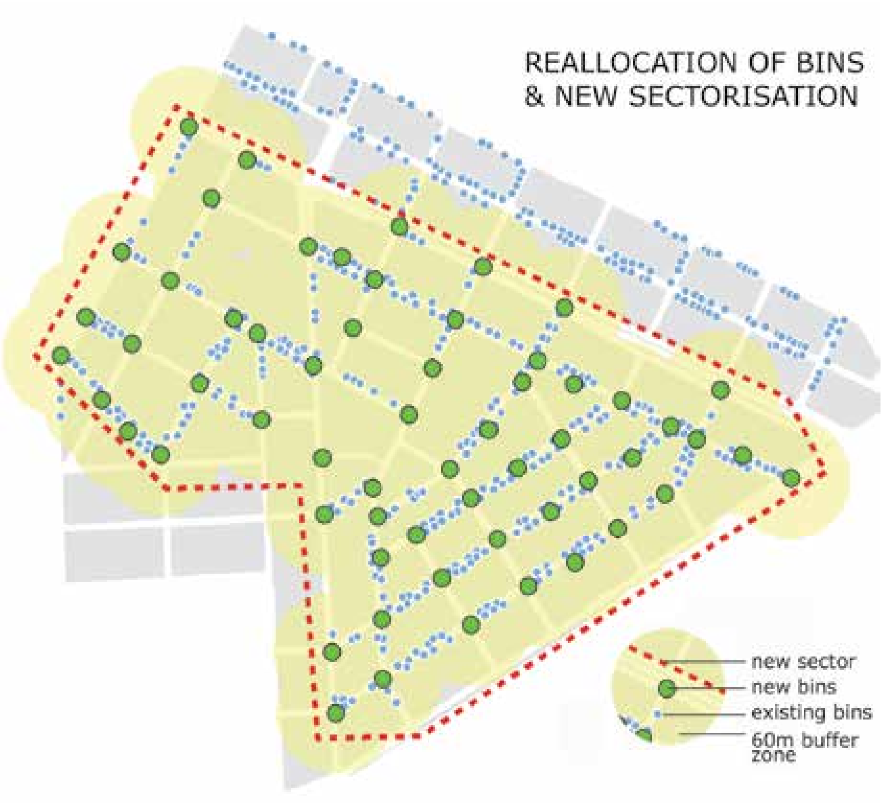

Fig. 5. Reallocation of waste collection bins in the new sector

The definition of the new sectors is restricted by the capacity of the available waste collection vehicles. Thus, the size (in terms of the number of bins) of a new sector was estimated at the 2/3 of the existing sector. Therefore, instead of 4 routes per week for each of the two subsectors (total: 8 routes per week) we designed smaller sectors and schedule 5 routes per week in these new sectors.

As a result of the above mentioned approach, each new sector should contain 95 bins, which can be collected in one vehicle trip. The reallocation of bins was based on travel distance from each residence to the nearest bin and the general intention to decrease the total number of bins. A maximum travel distance of 60 meters from each resident to the proposed new site of the bin was allowed. Moreover, the introduction of new bins with larger capacity, to accommodate for the same waste quantity, ensures the decrease in the total number of bins and collection stops. A higher priority for the allocation of the new bins was given to locations of bins in the existing system and to crossroads in order to facilitate social acceptance and collection vehicle travel.

Summarising, we assume a new waste collection planning: the MoN is divided into 22 new sectors and each collection vehicle should make 5 collections per week in each of these sectors. Thus we propose an improved collection schedule for the study area, as the vehicle collects each bin 5 times per week instead of 4, according to the existing situation. For this study we did not proceed to the full re-sectorisation for the total area of the municipality, but limited our approach within Sector1. Thus, we assumed a new sector (Sector_N1) within Sector1, with the properties described above (2/3 of the size of Sector 1, 5 collections per week). The evaluation of the results of the proposed modelling approach was based on the comparison between Sector_N1 and corresponding part of Sector1.

4.3.2 Routing – Network Analysis

After the reallocation of the waste collection bins and the definition of Sector_N1 the optimisation of waste collection vehicle routing was performed, using the ArcGIS Network Analyst modelling package. The optimal path finding algorithm of NA is an alteration of the classic Dijkstra's algorithm (Dijkstra, 1959) which solves the problem of optimal route selection on an undirected, nonnegative weighted graph in a reasonable computational time.

Fig. 6. Optimal waste collection route.

In the literature, many modifications and new algorithms have been used for the incorporation of the aforementioned restrictions. In the context of ArcGIS Network Analyst commercial GIS software, this algorithm is improved further, using effective data structures

such as d-heaps (ESRI, 2006). To use it within the context of real transportation data, this algorithm must be modified in order to respect real problem restrictions, such as one-way roads, prohibited turns (e.g. U-turns), demand at intersections (nodes) and along the roads, and side-of-street constraints while minimising a user-specified cost attribute. The key point is to build a cost matrix containing the costs between origins and destinations. These points correspond to pairs of vehicle stops (waste bins).
The total vehicle travel time is the sum of the travel time for each road segment plus the collection time for emptying of the bins. The user can define all the relevant traffic restrictions described above, the time delay for each stop for bin collection, as well as the first and last collection stop within the sector. The final output is the optimal solution in terms of distance or time criteria (fig. 6).

4.4 Results and discussion

The method described above was applied to simulate the waste collection procedure of the study area. Based on the methodology presented in the previous sections and the criteria and restrictions introduced in ArcGIS Network Analyst, different routing solutions were created for the collection of the new bins (95 bins of 1100 L) in their new location within Sector_N1. Evaluation of the results of the developed methodology is based on the comparison of the proposed waste collection scenario (Sp) with the existing one (Se). The time needed during waste collection has three distinct components: 1) time for hauling; (assumed as 25+25 km with average speed 50 km/h); 2) time for driving during collection; and, 3) time for emptying the bins.
The parameters input to the model were based on real data provided by the MoN and verified by field studies. More specifically, the time for emptying of the bins (bin loading, emptying and unloading – component 3) is 30 sec for bins with capacity up to 330 L and 60sec for bins with capacity equal to or larger than 660 L. The time for driving during collection (component 2) is determined by the average speed of the collection vehicle in the travel between stops and the total distance travelled in the collection segment of the route. For MoN the average speed is 5, 10 and 15 km/hr for 1-way, 2-way and central roads, respectively.
Both parameters are not readily available and default literature values are scarce. Sonesson (2000) reports values for the time required for bin emptying from empirical data for the wider Uppsala area in Sweden, as follows: 68.4 sec for inner city, 43.2 sec for suburbia and 57.4 sec for rural areas. Although the bin size is not defined, these values are in good agreement with the observed figures in the MoN. The author also reports an average collection speed of 20, 30 and 50 km/h for inner city, suburbs and rural areas, respectively. This is higher than the values achieved in MoN (conditions comparable with the inner city in Uppsala). Possible explanation is twofold: 1) different conditions of the road network and traffic in the two cities; and, 2) a denser matrix of collection points, due to a higher population density, allowing for shorter distances travelled between collection points and therefore lower speed. Nevertheless, the vehicle speed used for central roads in Nikea (15 km/h) compares well with the inner city collection speed in Uppsala (20 km/h).
The comparison of results, on a weekly basis, between the existing collection scenario (Se) and the proposed one (Sp) is illustrated in Table 3. The optimal solution expressed in Scenario Sp (Fig. 6) corresponds to 287 km of distance travelled by the waste collection vehicle on a weekly basis. This provides a 3% improvement when compared to the existing equivalent empirical route (Se). The improvement is more significant if assessed in terms of the total travel time in the optimal route, defined as the runtime of the collection vehicle

plus collection time for the waste bins. The total travel time, on a weekly basis, for the optimal route (Sp) is estimated to be 1225 minutes (18% reduction compared to the empirical route (Se). For the calculations the hauling time to the Fyli landfill (~25 km from Sector 1) should be added. Assuming an average speed of 50 km/h, the travel time to and from Fyli is about one hour.
Restricting the discussion to the collection phase only of the WC&T cycle, it is expected that fuel consumption relates more to time of operation and number of stops than distance travelled, as most of the collection time is spent for bin loading and emptying. Fuel consumption and corresponding gas emissions are functions of work performed for stopping and accelerating, actual driving, traffic related stops and lifting and compacting the waste (Sonesson 2000).

	Se	*Sp*	*Savings*
Distance (km)	296.5	287.5	9 (3.1%)
Time (h)	24.9	20.4	4.5 (18.1%)
V_{mean} (km/h)	11.9	14.1	2.2 (18.5%)
Fuel consumption (L)	266.9	230.0	36.9 (13.8%)
Cost (in €, 1L=1.4 E)	373.6	322.0	51.6 (13.8%)*
CO_2 (kg)	274.9	240.1	34.9 (12.7%)

Table 3. Comparison between the existing (*Se)* and the proposed (*Sp)* waste collection scenarios. (*) Cost savings are restricted to fuel costs and would be higher if maintenance and personnel costs were considered.

Therefore, even for the same distance travelled, changes in the number of stops, i.e. the number of the collected bins, can considerably affect fuel consumption and respectively, CO_2 emissions. In this study fuel consumption values and CO_2 emissions were calculated for heavy vehicles (8 – 16 tones) using the following formula (Hickman, 1999):

$$\varepsilon = \mathrm{K} + a \cdot v + b \cdot v^2 + c \cdot v^3 + \frac{d}{v} + \frac{e}{v^2} + \frac{f}{v^3} \tag{2}$$

where: ε is the emission value (gr/Km); K: constant value; a-f: coefficients; and, v: mean velocity of the vehicle (km/h).
The heavy dependence of collection time on the number of stops in combination with the new time schedule constitutes the main explanatory factor for the significant differences in the percentage savings in distance and time. Routing using the GIS modelling resulted to a 3.1% improvement of the distance travelled, although larger new sectors were proposed in comparison with the existing subsectors. In all the other values (fuel consumption, collection cost and CO_2 emissions, the percentage savings are estimated to exceed 10%. Finally, according to rough calculations, (extrapolation of the percentage savings to the total area of the municipality), the total savings for the municipality in one year, only from the reduction in fuel consumption, could approximate €68,000 and 46 tons CO_2 emissions, compared with the existing collection procedure.

5. Conclusions

GIS technology supports the optimisation of municipal solid waste management as it provides an efficient context for data capture, analysis and presentation. Two main categories of GIS-based waste management applications can be identified in the international literature. In the first, GIS is used for the selection of waste disposal landfills, and to a smaller extent, other waste treatment facilities. Most of these applications benefit from map overlay GIS functions and spatial allocation modelling methods. The final output of an application of that type is the suitability map of the area under investigation. This map could be the core of a spatial decision support system for a landfill site / waste treatment facility selection problem.

The second, more complex category of GIS supported waste management applications is related to waste collection. There are several applications for route optimisation, reallocation of waste bins and complete redesign of the collection sectors. The main aim of these applications is to reduce the collection distance and/or time of the collection vehicle fleet. The implementation of GIS-based modelling for waste collection optimisation in many countries with different socioeconomic conditions and technological background shows that significant savings could be achieved in most setups. The optimisation of routing has a direct positive impact on cost savings (reduction of fuel consumption and maintenance costs) as well as significant environmental impacts due to the lower levels of sound pollution within the urban environment and the reduction of greenhouse gases emissions. The application of GIS-based waste collection modelling should consider the following aspects, in order to provide reliable results:

a. Accurate and up to date information about the road network of the area under investigation.
b. Detailed capture of the spatial properties of the existing collection system (collection routes, location and attributes of waste bins, existing time schedule). Most often, especially in developing countries, the research team has to acquire this information with field work.
c. Installation of a modern GIS facility within the municipality enriched with network analysis functions. Advanced training of the staff is a very important factor for the efficient operation of this system.
d. Validation of the outputs from GIS-based modelling in order to ensure the applicability of the proposed routes in real life conditions.

Nowadays, although GIS-supported waste collection modelling is a mature scientific field the general diffusion of this technology is hampered by factors such as the absence and the poor quality of digital spatial data, the high cost of spatial data capture and the lack of personnel with the proper technological background to operate such modelling.

The methodology developed in this study and its application to the Municipality of Nikea, Athens, resulted in significant savings, especially in terms of time (18%), fuel consumption (13.8%) and CO_2 emissions (12.7%). The study demonstrated the value of GIS technology as a waste collection optimisation tool, capable of supporting decision making, in the context of a Mediterranean, densely populated city. The adoption of this technology could provide significant financial and environmental benefits for local communities.

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Comparison of the Suitability of Two LCA Procedures in Selecting the Best MSW Management System

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1. Introduction

Nowadays, the qualitative and quantitative assessment of the environmental impacts produced by every human activity is a topical field of research. In fact, all over the world a growing amount of attention is being given to the environmental issues and influences exerted by productive and management sectors. In particular, the management of waste is a crucial sector involving important aspects of our life, with it producing several environmental impacts that have to be adequately monitored and managed in a sustainable development perspective.

This cgapter focuses on the study of different municipal solid waste management systems in a district of the Campania region, in Southern Italy, which is sadly known due to it is suffering from a serious solid waste emergency that has lasted over 15 years. It has been the culmination of a process of insufficient implementation of European waste legislation for which Italy has repeatedly been condemned by the European Court of Justice. In particular, the images of heaps of rubbish in the streets of Naples and other nearby cities as well as the revolt of people against the realization of landfills and incinerators have been impressively documented by the international press (De Feo and Malvano, 2009; De Feo and De Gisi, 2010).

In order to manage these questionable situations, giving clear as well as affordable information to the people about the environmental impacts of waste management plants is fundamental. In this perspective, the study focused on the evaluation of the positive (induced) and negative (avoided) impacts caused on different environmental components by several municipal solid waste management systems defined on a provincial scale. This assessment was carried out by means of two different Life Cycle Assessment (LCA) procedures called WISARD and SimaPro, respectively.

LCA is a general methodological framework introduced to assess all the environmental impacts relating to a product, process or activity by identifying, quantifying and evaluating the overall resources consumed as well as all the emissions and wastes released into the environment (De Feo and Malvano, 2009).

In 1990, the society for environmental toxicology and chemistry (SETAC) defined the concept of LCA and developed a general methodology for the carrying out of LCA studies (Azapagic, 1999; De Feo and Malvano, 2009). The term ‘‘LCA’’ is used most frequently to describe all the cradle-to-grave approaches (Curran, 1996). A lot of these tools have been separately developed by different groups of specialists in order to support the decision maker within environmental management processes (SETAC, 1996; De Feo and Malvano, 2009). LCA methodology, as defined by SETAC or by ISO (International Organization for Standardization), consists of four steps (Curran, 1996; SETAC, 1996; De Feo and Malvano, 2009): (1) goal and scope definition, (2) inventory analysis, (3) impact assessment and (4) improvement assessment.

LCA can be useful and conveniently applied only to the life cycle related to the collection, treatment and landfill disposal of solid waste. In this particular case, the reference flux is given by the amount of waste produced by a community, while the output is represented by the emission of pollutants due to the several parts of the MSW management system. Therefore, the LCA procedure applied to the MSW management can be seen as a useful analysis instrument aimed at the evaluation of possible actions. In fact, the European Commission's Thematic Strategy on the prevention and recycling of waste outlines how adopting a life cycle perspective is essential for the sustainable management of wastes (Koneckny and Pennington, 2007; De Feo and Malvano, 2009).

The LCA procedures were able to calculate the consequences produced by the whole system as well as by each phase; in this case the goal is also to compare the two procedures. In particular, WISARD has been specifically designed for MSW applications, while SimaPro is a more general tool.

The aim of this study was to apply the two LCA procedures to MSW management on a provincial scale in order to choose the ''best" management system in environmental terms (impacts minimization) as well as compare the results obtained with a specific tool (WISARD), on one hand (the sociological and environmental goal), and with a more general tool (SimaPro), on the other (the technical goal).

2. Material and methods

2.1 Study area and reference data

The study area was the Province of Avellino in the Campania region, in Southern Italy, with a surface area of 2,792 km^2 and a population of 422,292 inhabitants (National Institute of Statistics, 1st January 2007). The total MSW production was 140,177,372 kg, the specific daily MSW production was around 0,9 kg/inhabitant/d, while the MSW composition was based on the presence of 42% of putrescibles, 30% of paper and cardboard, 14% of plastics, 8% of glass and 3% of metals and 1% of textiles (Table 1).

Fraction	Percentage (%)	Production (t/year)
Putrescibles (ex. Garden)	30%	42053.2116
Putrescibles (garden)	12%	16,821.2843
Paper and Cardboard	30%	42053.2116
Plastics	14%	19,624.8320
Glass	8%	11,214.1897
Metals	3%	4205.3211
Textiles	2%	2803.5474
Undersieve	1%	1401.7737
Total	100%	140,177.372

Table 1. MSW composition of the study area (De Feo and Malvano, 2009)

2.2 MSW management scenarios

The LCA study was developed considering twenty-one different MSW management scenarios. They were obtained considering different separated collection percentages, as well as various types of treatment for the dry residue deriving from the MSW without the materials being separated, collected and recycled or composted.

The MSW management scenarios considered can be conveniently divided into three categories: the first includes the scenarios from 1 to 10 (Fig. 1) and is based on the incineration of the dry residue ("Incineration scenarios"), the second includes the scenarios from 11 to 20 (Fig. 2) and is based on the sorting of the dry residue ("Sorting scenarios"), while the third relates only to scenario 21 (Fig. 3) and does not consider the treatment of dry residue, directly disposed of in landfill.

Scenarios 1-10 were based on a separated kerbside collection of paper and cardboard, putrescibles and dry residue, on a combined kerbside collection of plastics and metals and, finally, on a bring collection of glass with banks. The collected materials of paper and cardboard, glass, plastics and metals are then transported to recycling plants. Putrescibles, after collection, are transported to a composting plant. The dry residue is firstly transformed into RDF pressed bales and subsequently transported to an incineration plant. Discards deriving from all the treatment processes are collected and transported to a landfill. The ten scenarios (1–10) differ only in the percentage of separated collection. In fact, scenario 1 was based on a 35% separated collection, which was the lowest level allowed by Italian legislation, while scenario 10 was based on an 80% separated collection, a threshold which is difficult to achieve and only relates to a few and/or well organized territories. Scenarios 2–9 were progressively obtained by adding a 5% value to the separated collection of the previous scenario (Fig. 1).

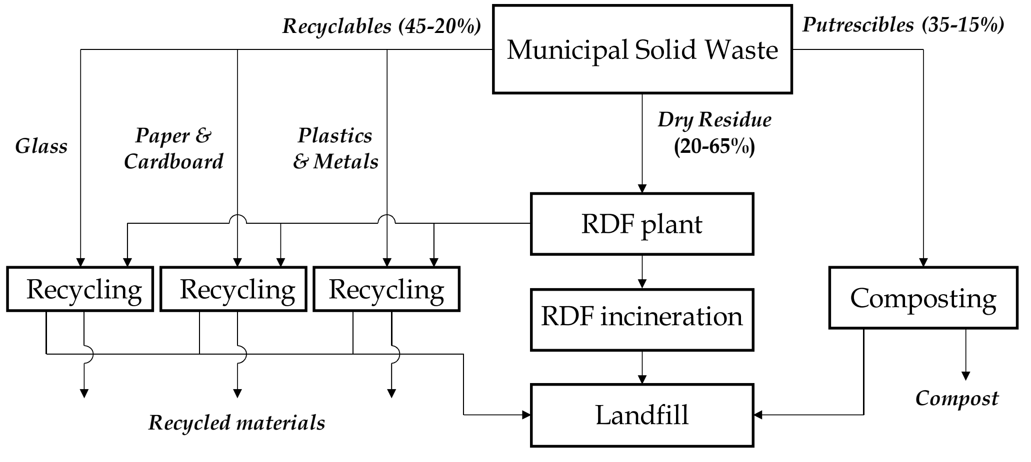

Fig. 1. Flow chart of MSW management scenarios 1–10 (De Feo and Malvano, 2009)

Scenarios 11-20 differ from scenarios 1-10 in only the treatment of the dry residue, which is transported to a sorting plant for a supplementary recovery of materials.

Management scenario 21 differs from scenario 20 (based on an 80% separated collection) with the dry residue being directly transported to a landfill (Fig. 3).

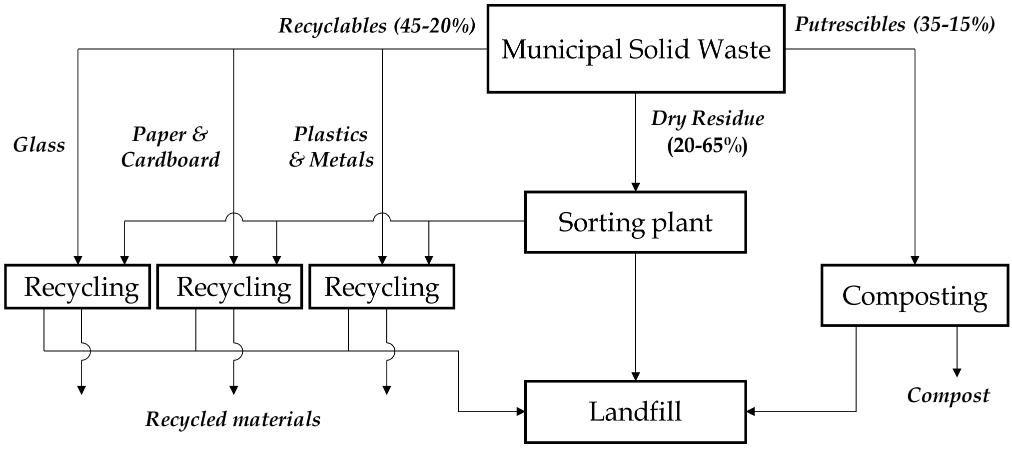

Fig. 2. Flow chart of MSW management scenarios 11-20 (modified by De Feo and Malvano, 2009)

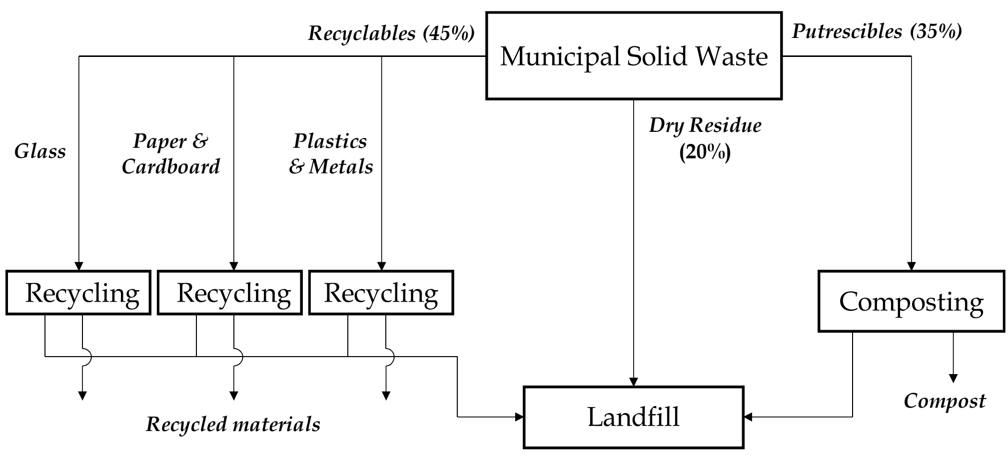

Fig. 3. Flow chart of MSW management scenario 21 (De Feo and Malvano, 2009)

2.3 The LCA procedures: SimaPro and WISARD

The Goal and Scope of the study was the use of LCA for the analysis of different MSW management systems in order to characterize the environmental impact as well as provide information to decision makers in choosing the best management solution to be adopted on a provincial level. The Function of the study was the activities of treatment and disposal of MSW, while the Functional Unit (quantified performance of a product system for use as a reference unit) was a tonne of waste of specific composition and, finally, the Reference Flow (measure of the outputs from processes in a given product system required to fulfil the function expressed by the functional unit) was quantified as the production of MSW in a year.

The Definition of Goal and Scope, Function, Functional Unit and Reference Flow allowed for the construction of the main MSW management system to be analyzed. The boundaries of the system were subsequently expanded to take into account the production processes avoided by energy recovery as well as recycling of matter. The operation of boundaries expansion is necessary in any LCA procedure in order to eliminate the potential environmental impacts that would be induced by the avoided processes of primary production (due to secondary production) from the results of the system analyzed.

The next step of the LCA procedure is the Inventory Analysis (compilation and quantification of inputs and outputs), which is the most important phase of the activity because it allows for the acquisition of all the information which is useful in compiling and quantifying the flows of matter and energy in input and output from each phase for the quantification of emissions.

Data about the Inventory Analysis of the WISARD procedure are reported in De Feo and Malvano (2009), which contains all the information pertaining to the mass and energy balances of the treatment plants of any MSW components. While, the full details of the Inventory Analysis of the SimaPro procedure are presented here.

The following modules, described in greater detail later, were implemented: Packaging Glass Green at Plant, Aluminium Secondary, from old scrap at Plant, Recycling Paper, with deinking at Plant, Recycling Plastics, Compost, at Plant, Glass Virgin, Aluminium Primary, at Plant, Thermomechanical Pulp, at Plan, PET, HDPE, LDPE, Ammonium Nitrate, Single Superphosphate, Potassium Sulphate, Landfill, Municipal Waste Incineration Plant, Wastewater Treatment Plant (PRé Consultants, 2007a, b).

The utilized data were deduced from average European plants as well as Italian specific plants that best approximate the systems to be adopted on a provincial level as well as best meet the requirements during the Goal and Scope definition of the study. The analysis was carried out on three levels. In fact, the Inventory was drawn up simultaneously taking into account:

- raw materials and energy used;
- transport of products, waste treatment and construction, dismantling and disposal of production sites;
- characterization of the machinery necessary for production and processing.

In particular, data were deduced from two principal sources: the Ecoinvent database and real data relating to MSW treatment and disposal plants operating in Italy and, particularly, in the Campania region. The MSW management model was constructed on the basis of several hypothesis, further verified with specific evaluation tests. In particular, assumptions were made in relation to the type of goods produced by "primary production" and "secondary production". Moreover, selection, recovery and recycling efficiencies for all types of materials were adopted. The basic assumption is that 1 kg of material produced by recycling replaces 1 kg of material produced from raw materials (Rigamonti et al., 2009).

Table 2 and 3 respectively show the type of packaging products and selection, recovery and recycling efficiencies adopted in the study.

Material	Primary Production	Secondary Production
Aluminium	Ingot	Ingot
Glass	Container	Container
Paper	Thermomechanical Paper	Pulp
Plastic	Grains of PET, HDPE, LDPE, LLDPE, PP	Grains of PET, HDPE, Mix (LDPE, LLDPE, PP)

Table 2. Type of packaging products (Rigamonti et al., 2009)

Material	Efficiency of Selection (%weight)	Efficiency of Recovery (%weight)	Efficiency of Recycling (%weight)
Aluminium	95	93	88.3
Glass	94	100	94
Paper	95	90	85.5
Plastic	80	73.5	58.7
Garden Waste	80	37.5	30

Table 3. Selection, recovery and recycling efficiencies (Rigamonti et al., 2009)

2.3.1 Composting plant

Putrescibles are treated by means of an aerobic composting process for the production of high quality compost to be used for farming in substitution of traditional chemical fertilizers. The basic assumption is that 1 kg of compost replaces a certain amount of artificial fertilizer so that the intake of nutrients N, P and K remains unchanged. A ton of compost contains: 6.2 kg N, 2.0 kg P and 4.5 kg K. Table 4 shows the general characteristics as well as consumption data of the composting plan t, useful for the Inventory Analysis. Energy required, type and quantity of polluting emissions as well as waste production relating to a treatment capacity of 10,000 tonnes of putrescibles per year. Moreover, they relate to a specific production of 1 kg of compost with a final water content of 50% by weight (Ecoinvent Data).

Composting Plant – Compost, at plant		
General Characteristics		
Life Time (year)	Treated Tonnes (t/m)	Type
10 – Stationarity Machinery 5 – Mobile Machinery 25 – Structural elements	10,000	Mechanized
Consumption		
Diesel (kg)	Electricity (kWh)	Water (l)
2.65E-3	1.18E-2	0

Table 4. Characteristics of the composting plant (modified by Nemecek et al., 2004)

2.3.2 Glass recycling plant

The recovery of glass was analyzed both in terms of preparation and selection of glass waste from separate collection as well as in terms of recycling activity (fusion, secondary packaging production, cooling, packaging and transporting to end users). The treated materials are crushed and selected by means of both manual and automatic processes with the removal of 100% of the impurities originally present. Table 5 shows the general characteristics as well as consumption data of the glass recycling plant, useful for the Inventory Analysis.

Glass Recycling Plant – Packaging Glass Green at plant		
General Characteristics		
Life Time (year)	Treated Tonnes (t/m)	Type
20 – Stationarity Machinery 5 – Mobile Machinery 50 – Structural Elements	100,000	Mechanized
Consumption		
Diesel (kg)	Electricity (kWh)	Water (l)
4.19E-2	2.44E-1	1.98E-3
Oil (MJ)	Natural Gas (MJ)	-
4.33E-2	3.57	-

Table 5. Characteristics of the glass recycling plant (modified by Hischier, 2007)

2.3.3 Paper recycling plant

The management of paper and cardboard waste involves the following phases: collecting, selecting and transporting to the recovery facilities. The recovery process considered was recycling without deinking with consumption of electricity and subsidiary materials, emission of pollutants into the air and wastewater treatment. Only natural gas was used as fuel for the heat production. While, a fuel mix of 16.1% coal, 70.3% methane and 13.6% fuel oil was used for electricity production. The recycling treatment was compared with the classical process of paper production from raw materials. The technology used is the thermal-mechanical treatment for the removal of fibres from wood chips. Table 6 shows the general characteristics as well as consumption data of the paper recycling plant, useful for the Inventory Analysis.

Paper Recycling Plant – Recycling Paper without deinking at plant		
General Characteristics		
Life Time (year)	Treated Tonnes (t/m)	Type
20 – Stationarity Machinery 5 – Mobile Machinery 50 – Structural Elements	33,000	Mechanized
Consumption		
Diesel (kg)	Diesel (kg)	Diesel (kg)
0.6555	7.9E-1	1.07E-2
Oil (MJ)	Natural Gas (MJ)	Coal (MJ)
0.6555	6.7769	1.552

Table 6. Characteristics of the paper recycling plant (modified by Hischier, 2007)

2.3.4 Aluminium recycling plant

Aluminium deriving from MSW separate collection is sent to facilities for the selection and subsequent recycling for the production of secondary aluminium products. The process is based on the use of “old” scrap deriving from separate collection and prepared by means of the selection and removal of organic matter in order to be suitable for the subsequent fusion process. The efficiency of recycling was assumed equal to 93%. The Life Cycle Analysis considers emissions from aluminium production from raw materials, as well. In particular, data from the Ecoinvent database and references concerning the best technologies used in industry for the production of non-ferrous metals are shown in Table 7.

Aluminium Recycling Plant – Aluminium Secondary, from old scrap at plant		
General Characteristics		
Life Time (year)	Life Time (year)	Life Time (year)
50	10,000	Mechanized
Consumption		
Oil (MJ)	Electricity (kWh)	Water (l)
5.13	2.88E-1	0
Natural Gas (MJ)	-	-
8.27	-	-

Table 7. Characteristics of an aluminium recycling plant (modified by Althaus et al., 2004)

2.3.5 Plastic recycling plant and mechanical–biological plant

In the Inventory Analysis developed for the waste treatment and disposal plants, the available data have allowed for a precise and detailed characterization of all the process units with the exception of those relating to the plants of plastics recycling plants and plants of mechanical and biological treatment (MBT) of dry residue as designed in the Campania region. For plastic recycling and MBT plants, in particular, the analysis only took into account the information relating to the consumption of matter and energy of the process, without considering the consumption of a second or third level related to the construction of the production site as well as production of machineries contained in the plants.

Tables 7, 8 and 9 show the summary data of the energy balance relating to plastic recycling and MBT plants, respectively.

Plastic Recycling Plant	
Plastics Selection	
Fuel (kWh/t)	Diesel (MJ/t)
26.6	84
PET Recovery	
Fuel (kWh/$t_{R\text{-}PET}$)	Methane (MJ/$t_{R\text{-}PET}$)
258	2500
HDPE Recovery	
Fuel (kWh/$t_{R\text{-}HDPE}$)	Methane (MJ/$t_{R\text{-}HDPE}$)
379	650

Table 8. Data of the energy balance relating to plastic recycling plants (Rigamonti et al, 2009)

3. Results and discussions

3.1 Summary of results obtained with WISARD

With the WISARD procedure, only scenarios 1-10, 20 and 21 were studied. The outputs from each option modelled were analysed under eleven environmental effect categories as suggested by the WISARD procedure with the aim of carrying out a synthetic study of the data available (Pricewaterhouse Coopers, 2006). The impact assessment categories

Mechanical –Biological Plant	
General Characteristics	
Polyethylene Film (kg)	Water (l)
1.6E-4	0.088
Wire (kg)	Electricity (MJ)
3.00E-4	0.051
Diesel (MJ)	
0.01	

Table 9. Data of the energy balance relating to MBT plants (Arena, 2003)

suggested are as follows: renewable energy use, non-renewable energy use, total energy use, water, suspended solids and oxydable matters index, mineral and quarried matters, greenhouse gases, acidification, eutrophication, hazardous waste, non-hazardous waste (De Feo and Malvano, 2009).

Attention was given to both measuring the overall impact due to the application of the entire MSW management system adopted, as well as the evaluation of the specific contribution produced by each phase of the MSW management system. In fact, each system was subdivided into the following sixteen phases: glass collection logistics (GCL), glass collection recycling (GCR), glass collection disposal (GCD), paper collection logistics (PaCL), paper collection recycling (PaCR), paper collection disposal (PaCD), plastics and metals collection logistics (Pl&MCL), plastics and metals collection recycling (Pl&MCR), plastics and metals collection disposal (Pl&MCD), putrescibles collection logistics (PCL), putrescibles collection composting (PCC), putrescibles collection disposal (PCD), dry residue collection logistics (DRCL), dry residue collection recycling (DRCR), dry residue collection RDF incineration (DRCI), and dry residue collection disposal (DRCD) (De Feo and Malvano, 2009).

Therefore, 192 management phases were considered (corresponding to the product of 16 phases for 12 scenarios), while 2112 single impact values were analysed and compared (corresponding to the product of 11 impact categories for 192 management phases). Moreover, 132 total impact values were analysed and compared (corresponding to the product of 11 impact categories and 12 management scenarios) (De Feo and Malvano, 2009).

The goal of the study was to evaluate the results obtained (values of avoided or produced impact) in order to highlight the most environmentally sound scenarios for each environmental impact category, as well as the trend associated with the percentage of separate collection (for the first ten MSW management scenarios), thus evaluating the positive and negative effects of recycling and/or composting (Table 10). The LCA software tool calculates impact values, performing mass and energy balances on the basis of the amount of waste to be treated. For scenarios 1–10, these quantities vary linearly with the percentage of separate collection and therefore the impact values for each management phase also vary in the same manner. Since the sum of the linear function is a linear function, the total impact values for each category also have to vary linearly. Moreover, for each impact category and MSW management scenario developed, the management phase with the greatest avoided impact (Table 11) and the management phase with the greatest produced impact (Table 12) were highlighted. Finally, scenarios 10, 20 and 21 were compared in order to highlight for which impact categories for high percentages of separate collection a management system based on recovery and recycling but without incineration would be preferable (De Feo and Malvano, 2009).

In summary, the following outcomes were obtained with the WISARD procedure (De Feo and Malvano, 2009):

- Scenario number 21 (80% separate collection, no RDF incineration, dry residue sorting) was the most environmentally sound option for the following six impact categories: renewable energy use, total energy use, water, suspended solids and oxydable matters index, eutrophication, and hazardous waste;
- Scenario number 10 (80% separate collection, RDF production and incineration) was the most environmentally sound option for the following three impact categories: non-renewable energy use, greenhouse gases, and acidification;
- Scenario number 1 (35% separate collection, RDF production and incineration) was the most environmentally sound option for the following two impact categories: mineral and quarried matters, and non-hazardous waste;
- For the following eight impact categories (of the eleven considered), all the MSW management scenarios considered produced negative impacts, and the highest percentage of separate collection corresponded to the highest avoided impact: Renewable Energy Use, Non-Renewable Energy Use, Total Energy Use, Water, Suspended Solids and Oxydable Matters Index, Acidification, Eutrophication, and Hazardous Waste;
- For ''Mineral and Quarried Matters'' the MSW management scenarios considered produced positive and negative impacts, and the highest percentage of separate collection corresponded to the highest produced impact;
- For ''Greenhouse Gases'', the MSW management scenarios considered produced positive and negative impacts, and the highest percentage of separate collection corresponded to the highest avoided impact;
- For ''Non-Hazardous Waste'' all the MSW management scenarios considered produced positive impacts, and the highest percentage of separate collection corresponded to the highest produced impact;
- For the following six impact categories (of the eleven considered), for high percentages of separate collection (80%), a management system based on recovery and recycling but without incineration would be preferable: Renewable Energy Use, Total Energy Use, Water, Suspended Solids and Oxydable Matters Index, Eutrophication and Hazardous Waste;
- ''Paper Collection Recycling'' was the system component with the greatest avoided impact for 45.5% of the cases considered;
- ''Dry Residue Collection Logistic'' was the system component with the greatest produced for 54.5% of the cases considered.

3.1 Results obtained with SimaPro

The results obtained with the SimaPro procedure were evaluated by means of three keys.

The first key evaluates the results of the Inventory Analysis consisting of the data on the emissions of pollutants into the environment due to the different phases of the MSW management system, focusing on the treatment activities of the several MSW components. Thus, it was possible to compare in quantitative environmental terms, the impacts generated

Impact Category	MSW Management Scenario											
	35% (1)	40% (2)	45% (3)	50% (4)	55% (5)	60% (6)	65% (7)	70% (8)	75% (9)	80% (10)	80%S (20)	80%L (21)
Renewable Energy Use (equivalent inhabitant)	-428,695	-489,507	-543,589	-597,599	-650,858	-704,852	-725,547	-792,498	-860,128	-927,731	-952,854	-916,814
	Impact (equivalent inhabitant)= -10,628 × (percentage of separate collection) – 60,995										-2.7%	+1.2%
Non-Renewable Energy Use (equivalent inhabitant)	-14,934	-15,927	-17,227	-18,149	-18,912	-20,158	-21,379	-22,249	-23,494	-24,284	-23,655	-20,679
	Impact (equivalent inhabitant)= -209.89 × (percentage of separate collection) – 7,602										+2.6%	+14.8%
Total Energy Use (equivalent inhabitant)	-30,725	-33,002	-37,316	-40,264	-43,030	46,288	-48,253	-51,646	-55,425	-58,764	-59,117	-54,880
	Impact (equivalent inhabitant)= -607.49 × (percentage of separate collection) – 9,640										-0.6%	+6.6%
Water (equivalent inhabitant)	-2,991	-3,589	-4,191	-4,785	-5,568	-6,168	-6,784	-7,388	-8,174	-8,756	-9,117	-8,876
	Impact (equivalent inhabitant)= -129.18 × (percentage of separate collection) – 1,588										-4.1%	-1.4%
Suspended Solids and oxydable matters index (equivalent inhabitant)	-6,905	-7,905	-8,945	-10,075	-11,075	-12,031	-13,039	-14,044	-15,045	-15,894	-16,393	-15,874
	Impact (equivalent inhabitant)= -201.53 × (percentage of separate collection) – 96,36										-3.1%	0.1
Minerals and Quarried Matters (t)	-1,279	-767	51	257	1,077	1,920	2,712	3,481	3,504	4,375	21,396	27,404
	Impact (t)= 129.28 × (percentage of separate collection) – 5,910										+389.1%	+526.4%
Greenhouse gases (equivalent inhabitant)	58	-677	-108	-788	-969	-1,466	-1,444	-1,899	-2,077	-2,337	4,370	10,425
	Impact (equivalent inhabitant)= -51.85 × (percentage of separate collection) + 1,810										+287%	+546.1%
Acidification (equivalent inhabitant)	-8,968	-9,247	-9,600	-9,871	-9,986	-10,337	-10,603	-10,710	-11,148	-11,321	-10,296	-4,237
	Impact (equivalent inhabitant)= -51.61 × (percentage of separate collection) + ,211= 0.9931										+9.1%	+62.6%
Eutrophication (equivalent inhabitant)	-1,120	-1,291	-1,460	-1,634	-1,806	-1,979	-2,152	-2,314	-2,486	-2,593	-2,619	-2,517
	Impact (equivalent inhabitant)= -33.470 × (percentage of separate collection) + 40,8										-1.0%	+2.9%
Hazardous waste (t)	-1,375	-1,427	-1,480	-1,543	-1,565	-1,617	-1,659	-1,702	-1,755	-1,807	-2,204	-1,642
	Impact (t)= -9.32 × (percentage of separate collection) + 1,056										-22.0%	+9.1%
Non Hazardous Waste (t)	21,023	21,477	22,038	22,511	23,065	23,676	24,263	24,732	25,387	25,824	42,317	43,470
	Impact (t)= 108.89 × (percentage of separate collection) + 17,139										+63.9%	+68.3%

Table 10. Summary of the numerical results obtained with WISARD for MSW management scenarios 1-10 developed in terms of avoided or produced impact (De Feo and Malvano, 2009)

Impact Category	MSW Management Scenario											
	35% (1)	40% (2)	45% (3)	50% (4)	55% (5)	60% (6)	65% (7)	70% (8)	75% (9)	80% (10)	80% s (20)	80% l (21)
Renewable Energy Use (equivalent inhabitant)	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR
	-380,753	-441,945	-496,339	-550,732	-605,125	-659,518	-679,913	-747,908	-815,899	-883,891	-883,891	-883,891
Non-Renewable Energy Use (equivalent inhabitant)	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR
	-10,791	-12,140	-13,759	-15,378	-16,727	-18,345	-19,694	-21,313	-22,932	-24,281	-24,281	-24,281
Total Energy Use (equivalent inhabitant)	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR
	-14,973	-17,360	-19,514	-21,641	-23,795	-25,923	-26,779	-29,426	-32,073	-34,746	-34,746	-34,746
Water (equivalent inhabitant)	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR
	-4,201	-4,749	-5,297	-5,845	-6,575	-7,123	-7,671	-8,219	-8,950	-9,498	-9,498	-9,498
Suspended Solids and oxydable matters index (equivalent inhabitant)	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR
	-6,825	-7,807	-8,821	-9,929	-10,863	-11,845	-12,827	-13,809	-14,791	-15,614	-15,614	-15,614
Minerals and Quarried Matters (t)	DRCR	DRCR	DRCR	DRCR	GCR	GCR	GCR	GCR	GCR	GCR	GCR	GCR
	-7,200	-6,622	-6,043	-5,575	-5,490	-5,948	-6,446	-6,954	-7,452	-7,951	-7,951	-7,951
Greenhouse gases (equivalent inhabitant)	DRCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR
	-11,152	-12,280	-13,518	-15,346	-16,583	-18,416	-19,667	-21,499	-22,736	-24,569	-24,569	-24,569
Acidification (equivalent inhabitant)	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR
	-16,488	-18,641	-20,795	-22,948	-25,101	-27,255	-29,408	-31,561	-33,715	-35,868	-35,868	-35,868
Eutrophication (equivalent inhabitant)	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR	PaCR
	-1,154	-1,328	-1,499	-1,675	-1,851	-2,029	-2,205	-2,369	-2,544	-2,655	-2,655	-2,655
Hazardous waste (t)	DRCR	DRCR	DRCR	DRCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR	Pl&MCR
	-1,938	-1,791	-1,613	-1,496	-1,544	-1,686	-1,829	-1,971	-2,114	-2,256	-2,256	-2,256
Non Hazardous Waste (t)	PCC	PCC	PCC	PCC	PCC	PCC	PCC	PCC	PCC	PCC	PCC	PCC
	-1,332	-1,437	-1,642	-1,846	-2,050	-2,155	-2,359	-2,564	-2,769	-2,974	-2,974	-2,974

Table 11. Management phase with the greatest avoided impact for each impact category and for MSW management scenarios 1-10 developed in the study performed with WISARD. DRCL = dry residue collection logistics; DRCD = dry residue collection disposal; DRCR = dry residue collection recycling; PaCR = paper collection recycling; Pl&MCR = plastics and metals collection recycling; GCR = glass collection recycling; PCC = putrescibles collection composting; PCD = putrescibles collection disposal (De Feo and Malvano, 2009)

	35% (1)	40% (2)	45% (3)	50% (4)	55% (5)	60% (6)	65% (7)	70% (8)	75% (9)	80% (10)	80%S (20)	80%L (21)
Renewable Energy Use (equivalent inhabitant)	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	GCL	GCL	GCL
	1,020	952	816	748	680	626	558	483	394	347	347	347
Non-Renewable Energy Use (equivalent inhabitant)	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL
	5,935	5,666	5,126	4,586	4,317	3,777	3,237	2,968	2,374	2,023	2,023	2,023
Total Energy Use (equivalent inhabitant)	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL
	5,748	5,486	4,961	4,440	4,178	3,657	3,135	2,873	2,299	1,959	1,959	1,959
Water (equivalent inhabitant)	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL
	676	639	584	530	493	438	384	329	274	237	237	237
Suspended Solids and oxydable matters index (equivalent inhabitant)	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	PCD	DRCD	DRCD
	41	38	35	32	29	26	23	20	16	17	21	24
Minerals and Quarried Matters (t)	DRCD	DRCD	PCD	PCD	PCD	PCD	PCD	PCD	PCD	PCD	DRCD	DRCD
	9,412	8,711	8,510	9,412	10,413	11,414	12,615	13,716	13,818	15,019	18,424	21,727
Greenhouse gases (equivalent inhabitant)	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	PCD	PCD	PCD	DRCD	DRCD
	10,003	9,269	8,535	7,828	6,929	6,222	5,502	5,415	5,832	6,249	8,053	10,527
Acidification (equivalent inhabitant)	DRCL	DRCL	DRCL	PCC	PCC	PCC	PCC	PCC	PCC	PCC	PCC	PCC
	17,001	15,428	14,587	14,702	16,046	17,596	18,950	21,294	21,248	23,198	23,198	23,198
Eutrophication (equivalent inhabitant)	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	DRCL	PCD	PCD	DRCD	DRCD
	69	65	58	53	49	44	39	34	37	39	49	55
Hazardous waste (t)	DRCI	DRCI	DRCI	DRCI	DRCI	DRCI	DRCI	DRCI	PaCR	PaCR	PaCR	PaCR
	1,262	1,165	1,068	970	903	806	719	631	580	610	610	610
Non Hazardous Waste (t)	DRCD	DRCD	PCD	PCD	PCD	PCD	PCD	PCD	PCD	PCD	DRCD	DRCD
	11,020	10,019	9,219	10,021	11,023	12,024	13,026	14,028	15,030	16,032	20,040	21,046

Table 12. Management phase with the greatest produced impact for each impact category and for MSW management scenarios 1-10 developed in the study performed with WISARD. DRCD = dry residue collection disposal; DRCL = dry residue collection logistics; DRCI = dry residue collection RDF incineration; DRCR = dry residue collection recycling; GCL = glass collection logistics; PaCR = paper collection recycling; Pl&MCR = plastics and metals collection recycling; PCC = putrescibles collection composting; PCD = putrescibles collection disposal (De Feo and Malvano, 2009)

by the production units of materials from raw materials and impacts resulting from treatment processes that lead to the production of secondary materials deriving from the separate collection.

The second interpretation key directly derives from the evaluation model adopted, which allows for the definition of the damage level induced by the MSW management system with reference to the following macro-categories: Human Health, Ecosystem Quality and Resource Consumption. Thus, it was possible to compare different scenarios and express judgments about the influence of the percentage of separate collection on the impacts produced. In particular, the damage category "Human Health" includes the following damage/impact sub-categories: Carcinogens, Respiration Organics, Respiration Inorganics, Climate Change, Radiation, Ozone Layer. While, "Ecosystem Quality" is the combination of data related to the following damage/impact sub-categories: Ecotoxicity, Acidification/Eutrophication, Land Use. Finally, "Resources consumption" comprises the sub-categories Minerals and Fossil Fuels.

The third and final key relates to the identification of the management phases having a significant impact on the overall impact as well as how these results vary with the scenarios considered.

3.1.1 Results of the inventory analysis

The analysis of the emission data related to the packaging materials highlighted that, in most cases, the pollutant emissions from secondary production were lower than that for primary production for each impact category. Tables 13, 14 and 15 show the results obtained for the packaging materials of glass, aluminium and paper, respectively.

Emissions	Primary Production	Secondary Production
CO_2	955 g	880.9 g
CO	1.42 g	0.825 g
NO_X	1.43 g	3.24 g
SO_X	5.07 g	4.85 g
BOD_5	0.584 mg	1.74 g
COD	0.011.9 g	2.18 g
Tot. Nitrogen	11.5 mg	10.1 mg
Sand	562 g	1.99 mg

Table 13. Comparison between the emissions due to the primary production of glass and recycling of the same quantity of glass (secondary production)

Emissions	Primary Production	Secondary Production
Dust (< 2.5 μm)	4.97 g	269 mg
Dust (> 10 μm)	12.3 g	622 mg
Dust (> 2.5 μm <10 μm)	7.43 g	232 mg
NO_x	19.8 g	2.58 g
Cadmium	628 μm	243 μm
BOD_5	20.7 g	1.86 g
COD	33.4 g	4.07 g
PAH	424 μm	23.4 μm
Chrome VI	18.9 mg	4.36 mg

Table 14. Comparison between the emissions due to the primary production of aluminium and recycling of the same quantity of aluminium (secondary production)

Emissions	Primary Production	Secondary Production
Water	16.8 m^3	590 l
Wood	1.2 mm^3	2.45 mm^3
CO_2	856 g	809.6 g
CO	586.4 mg	593.6 mg
Chrome VI	11 μm	15.9 μm
BOD_5	1.38 g	647 mg
Chlorine	3.96 g	3.73 g
COD	5.05 g	1.94 g
Mercury	11.5 μm	5.04 μm
Suspended Solid	1.35 g	308 mg

Table 15. Comparison between the emissions due to the primary production of paper and recycling of the same quantity of paper (secondary production)

The presentation of the results of the Impact Assessment in terms of Environmental Damage makes it possible to analyze the problem of potential impacts in general terms. While, it is subsequently possible to extrapolate more peculiar considerations (PRè Consultants, 2000).
Figures 4, 5 and 6 show the differences between the impact of secondary and primary production of glass, aluminium, paper and compost, for the damage categories Human Health, Ecosystem Quality and Resource Consumption, respectively. A positive value of the difference indicates an induced impact. Thus, for glass and paper the recycling process induce impacts both in terms of Human Health and Resource Consumption.

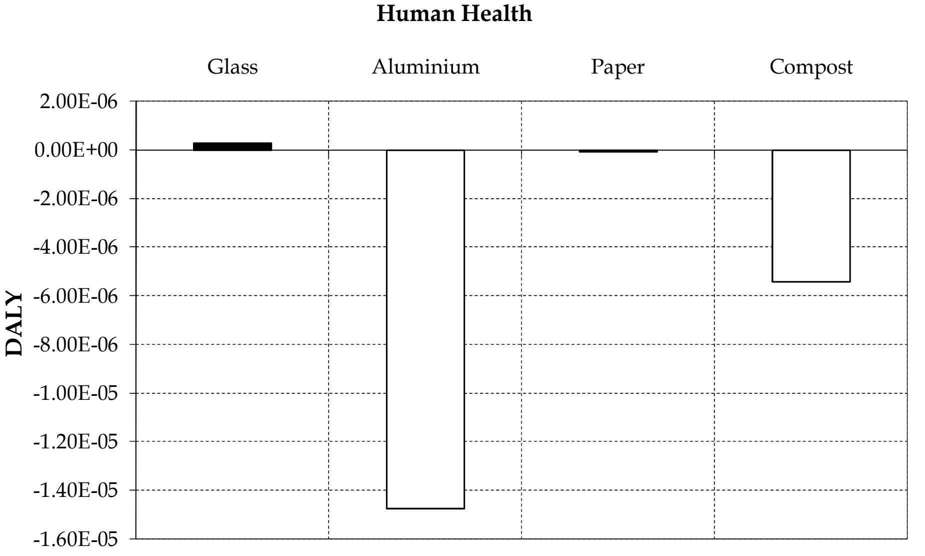

Fig. 4. Difference between impact due to primary production and secondary production of packaging materials and compost in terms of "Human Health" damage category (the disability-adjusted life year, DALY, is a measure of overall disease burden, expressed as the number of years lost due to ill-health, disability or early death)

In general, identical to the results obtained with WISARD, with reference to all the management scenarios considered it was highlighted that the environmental impact linearly decreases with the percentage of separate collection for each damage category. Only the subcategory "Acidification/Eutrophication" of the damage macro-category "Ecosystem Quality" showed an induced impact increasing with the percentage of separate collection (Table 16). Moreover, the MSW management system determines avoided impacts for the damage categories "Human Health" and "Resources Consumption", while it determines induced impacts for the damage category "Ecosystem Quality".
Taking into account the contribution of the different MSW management phases, it was noted that all the considered scenarios have negative impact indicators in terms of Human Health and Resource Consumption, thus indirectly indicating that in these cases an integrated management of MSW is more environmentally sound than traditional methods of production of materials and energy. Dry residue incineration, landfill disposal, composting and glass production were the MSW management phases with the greatest influence on the final results in terms of environmental impacts.

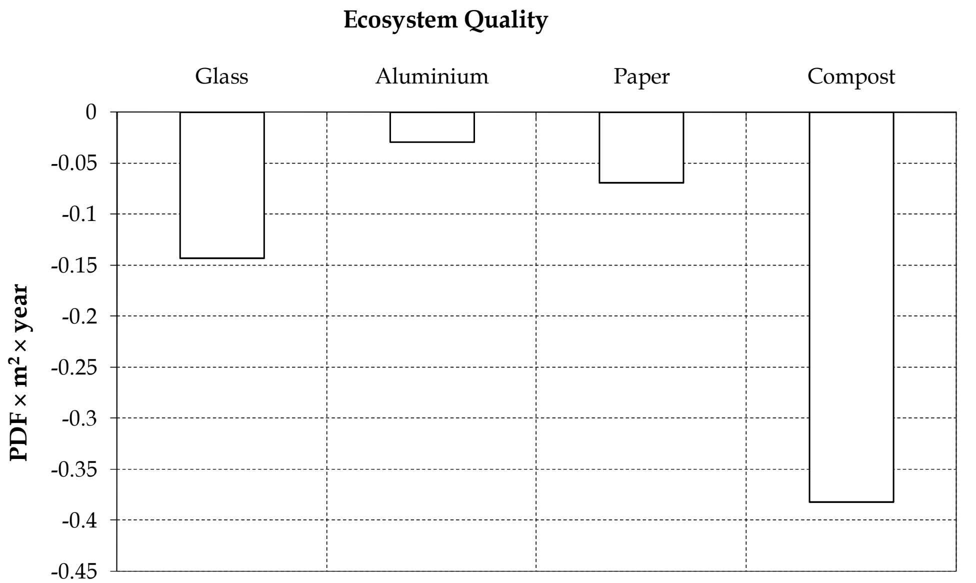

Fig. 5. Difference between impact due to primary production and secondary production of packaging materials and compost in terms of "Ecosystem Quality" (the Potentially Disappeared Fraction, PDF, is the fraction of species that has a high probability of no occurrence in a region due to unfavorable conditions)

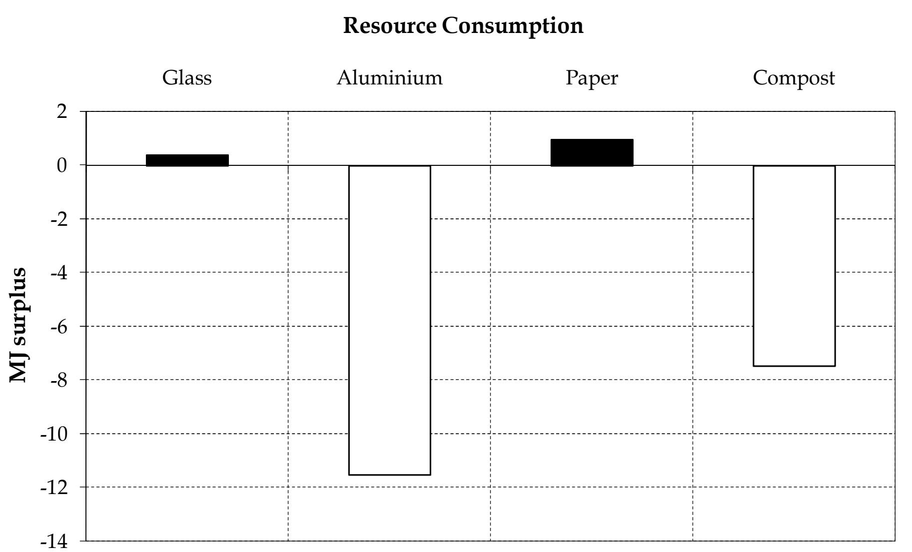

Fig. 6. Difference between impact due to primary production and secondary production of packaging materials and compost in terms of "Resource Consumption" damage category (MJ surplus expresses the surplus of Mega Joule needed in the extraction of resources when the demand for these will be 5 times higher than it was in 1990)

Due to the simplified basic hypothesis of the adopted model, the modality of plastics management was not considered when assessing the produced effects. Moreover, the modality of paper management did not compare with the quantitative analysis of results, with the analysis of single emissions highlighting that the balance between primary and secondary production is essentially neutral. Finally, the implemented model was not sufficiently adequate for the collecting and transporting phase, which would require the implementation of another calculation model.

Impact Category	MSW management scenario									
	35% (1)	40% (2)	45% (3)	50% (4)	55% (5)	60% (6)	65% (7)	70% (8)	75% (9)	80% (10)
Carcinogens (DALY)	-9,7E-05	-1,0E-04	-1,0E-04	-1,1E-04	-1,1E-04	-1,1E-04	-1,2E-04	-1,2E-04	-1,3E-04	-1,3E-04
(-), decreasing	Impact (Daly)= -7×10^{-7} × (percentage of separate collection) – 7×10^{-5}									
Resp. Organics (DALY)	-3,6E-07	-3,9E-07	-4,2E-07	-4,5E-07	-4,8E-07	-5,2E-07	-5,5E-07	-5,8E-07	-6,1E-07	-6,4E-07
(-), decreasing	Impact (Daly)= -6×10^{-9} × (percentage of separate collection) – 1×10^{-7}									
Resp. Inorganics (DALY)	-4,2E-04	-4,2E-04	-4,3E-04	-4,3E-04	-4,4E-04	-4,4E-04	-4,5E-04	-4,5E-04	-4,6E-04	-4,6E-04
(-), decreasing	Impact (Daly)= -1×10^{-6} × (percentage of separate collection) – 0.0004									
Climate Change (DALY)	-6,3E-05	-6,5E-05	-6,7E-05	-6,8E-05	-7,0E-05	-7,2E-05	-7,4E-05	-7,6E-05	-7,8E-05	-8,0E-05
(-), decreasing	Impact (Daly)= -4×10^{-7} × (percentage of separate collection) – 5×10^{-5}									
Radiation (DALY)	-1,7E-06	-1,9E-06	-2,0E-06	-2,2E-06	-2,4E-06	-2,6E-06	-2,7E-06	-2,9E-06	-3,1E-06	-3,3E-06
(-), decreasing	Impact (Daly)= -3×10^{-8} × (percentage of separate collection) – 5×10^{-7}									
Ozone Layer (DALY)	-6,5E-08	-7,0E-08	-7,5E-08	-8,0E-08	-8,5E-08	-9,0E-08	-9,5E-08	-1,0E-07	-1,0E-07	-1,1E-07
(-), decreasing	Impact (Daly)= -1×10^{-9} × (percentage of separate collection) – 3×10^{-8}									
Ecotoxicity (PDF × m^2 × yr)	8,3E+02	7,5E+02	6,7E+02	5,9E+02	5,1E+02	4,3E+02	3,5E+02	2,7E+02	1,9E+02	1,2E+02
(+), decreasing	Impact (PDF × m^2× yr)= -15.784 × (percentage of separate collection) + 1378.1									
Acidif/ Eutroph. (PDF × m^2 × yr)	-4,7E+00	-4,3E+00	-3,9E+00	-3,4E+00	-3,0E+00	-2,6E+00	-2,1E+00	-1,7E+00	-1,3E+00	-8,3E-01
(-), increasing	Impact (PDF × m^2 × yr)= 0.0872 × (percentage of separate collection) – 7.8006									
Land Use (PDF × m^2 × yr)	-5,0E-01	-6,0E-01	-6,9E-01	-7,9E-01	-8,8E-01	-9,8E-01	-1,1E+00	-1,2E+00	-1,3E+00	-1,4E+00
(-), decreasing	Impact (PDF × m^2 × yr)= - 0.0192 × (percentage of separate collection) + 0.1711									
Minerals (MJsurplus)	-5,0E+01	-5,5E+01	-6,0E+01	-6,5E+01	-6,9E+01	-7,4E+01	-7,9E+01	-8,4E+01	-8,9E+01	-9,4E+01
(-), decreasing	Impact (MJsurplus)= - 0.9713 × (percentage of separate collection) – 15.988									
Fossil Fuels (MJSurplus)	-3,3E+02	-3,6E+02	-3,8E+02	-4,1E+02	-4,3E+02	-4,6E+02	-4,8E+02	-5,1E+02	-5,3E+02	-5,6E+02
(-), decreasing	Impact (MJsurplus)= - 5.0632 × (percentage of separate collection) – 155.12									

Table 16. Summary of the numerical results obtained with SimaPro for MSW management scenarios 1-10 developed in terms of avoided or produced impact. (-) = avoided impact, (+) = induced impact. Decreasing = the avoided or induced impact decreases with the increasing of separate collection percentage; Increasing = the avoided or induced impact increases with the increasing of separate collection percentage.

Impact Category	MSW management scenario									
	35% (1)	40% (2)	45% (3)	50% (4)	55% (5)	60% (6)	65% (7)	70% (8)	75% (9)	80% (10)
Carcinogens	Thermal treatment	Thermal treatment	Thermal treatment	Thermal treatment	Thermal treatment	Thermal treatment	Thermal treatment	Thermal treatment	Thermal treatment	Wastewater treatment
Resp. Organics	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)
Resp. Inorganics	Glass recycling	Glass recycling	Glass recycling	Glass recycling	Glass recycling	Glass recycling	Glass recycling	Glass recycling	Glass recycling	Incineration
Climate Change	Incineration	Incineration	Incineration	Incineration	Incineration	Incineration	Incineration	Incineration	Incineration	Incineration
Radiation	Electricity consumption (nuclear)	Electricity consumption (nuclear)	Titanium dioxide production	Titanium dioxide production	Titanium dioxide production	Titanium dioxide production	Titanium dioxide production	Titanium dioxide production	Titanium dioxide production	Titanium dioxide production
Ozone Layer	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)
Ecotoxicity	Incineration	Incineration	Incineration	Incineration	Incineration	Incineration	Incineration	Incineration	Incineration	Incineration
Acidif/ Eutroph.	Composting	Composting	Composting	Composting	Composting	Composting	Composting	Composting	Composting	Composting
Land Use	Natural fertilizers	Natural fertilizers	Natural fertilizers	Natural fertilizers	Natural fertilizers	Natural fertilizers	Natural fertilizers	Natural fertilizers	Natural fertilizers	Natural fertilizers
Minerals	Titanium dioxide production	Titanium dioxide production	Titanium dioxide production	Titanium dioxide production	Titanium dioxide production	Titanium dioxide production	Titanium dioxide production	Titanium dioxide production	Titanium dioxide production	Titanium dioxide production
Fossil Fuels	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)	Glass (green)

Table 17. Management phase with the greatest produced impact for each impact category and for MSW management scenarios 1-10 developed in the study performed with SimaPro

	35% (1)	40% (2)	45% (3)	50% (4)	55% (5)	60% (6)	65% (7)	70% (8)	75% (9)	80% (10)
Carcinogens	Electricity consumption	Electricity consumption	Electricity consumption	Electricity consumption	Leachate disposal	Leachate disposal	Leachate disposal	Leachate disposal	Leachate disposal	Leachate disposal
Resp. Organics	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)
Resp. Inorganics	Electricity consumption	Electricity consumption	Electricity consumption	Electricity consumption	Electricity consumption	Electricity consumption	Glass (white)	Glass (white)	Glass (white)	Glass (white)
Climate Change	Electricity consumption	Electricity consumption	Electricity consumption	Electricity consumption	Electricity consumption	Electricity consumption	Glass (white)	Glass (white)	Glass (white)	Glass (white)
Radiation	Radioactive emissions	Radioactive emissions	Radioactive emissions	Radioactive emissions	Radioactive emissions	Radioactive emissions	Radioactive emissions	Radioactive emissions	Radioactive emissions	Radioactive emissions
Ozone Layer	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)
Ecotoxicity	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)
Acid./ Eutr.	Electricity consumption	Electricity consumption	Electricity consumption	Electricity consumption	Electricity consumption	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)
Land Use	Softwood	Softwood	Softwood	Softwood	Softwood	Softwood	Softwood	Softwood	Softwood	Softwood
Minerals	Bauxite consumption	Bauxite consumption	Bauxite consumption	Bauxite consumption	Bauxite consumption	Bauxite consumption	Bauxite consumption	Bauxite consumption	Bauxite consumption	Bauxite consumption
Fossil Fuels	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)	Glass (white)

Table 18. Management phase with the greatest avoided impact for each impact category and for MSW management scenarios 1-10 developed in the study performed with SimaPro

Table 17 indicates the management phase with the greatest produced impact for each impact category as well as for MSW management scenarios 1-10 developed in the study performed with SimaPro. "Glass (green)" resulted the heaviest phase 33 times out of 110 (10 scenarios x 11 impact categories), corresponding to 27.3%. While, "Incineration" and "Titanium dioxide production" were the heaviest phase 20 times (18.2%) and 18 times (16.4%), respectively.
Table 18 indicates the management phase with the greatest avoided impact for each impact category as well as for MSW management scenarios 1-10 developed in the study performed with SimaPro. "Glass (white)" resulted the lightest phase 53 times out of 110, corresponding to 48.2%. While, "Electricity consumption" was the heaviest phase 21 times (19.1%). Finally, "Radioactive emissions", "Softwood", "Bauxite consumption" were the lightest phase 10 times each one (9.1%).
More detailed results in terms of impacts due to the several phases of the MSW management system are presented in the next paragraphs in relation to the most significant impact categories.

3.1.2 Climate change

The impact produced by dry residue incineration decreased linearly with the increasing of the percentage of separate collection in terms of Climatic Change. A similar result was obtained by Bruno et al. (2002), also indicating that the solution with incineration was more environmentally sound than the solution with direct landfill disposal in terms of Acidification and Global Warming. Eriksson et al. (2005) identified in the incineration the management phase with the maximum production of CO_2, while waste landfilling was indicated as the worst option. The composting process of putrescibles was the management phase which affected the most the production of induced impacts. The impact increases linearly with the increasing of the percentage of separate collection. Arena et al. (2003) pointed out that the worst solution was the direct landfill disposal in terms of Climate Change, due to the emission of greenhouse gases, accordingly to the findings of Ozeler et al. (2006). For the scenario with 70% of separate collection, the impact induced by the composting process recycling overcame the impact induced by the dry residue incineration, in terms of Climate Change damage category (Figure 7). A similar solution was found by Bruno et al., (2002).

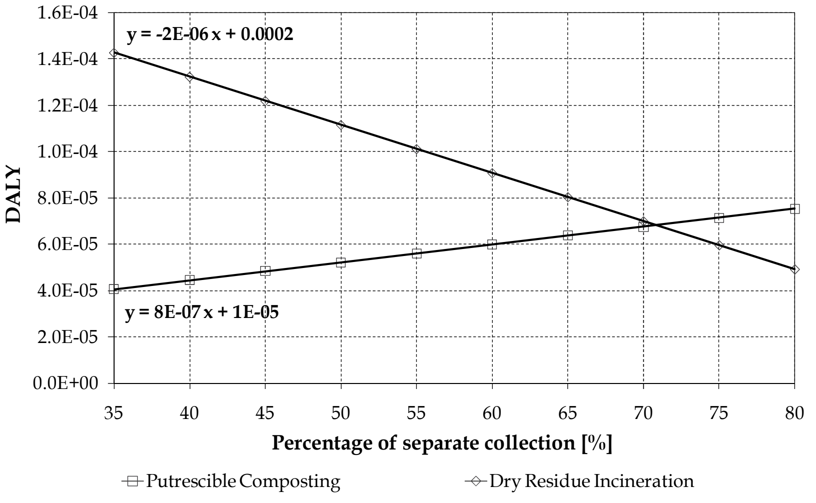

Fig. 7. Trend of the induced impacts by Putrescibles Composting and Dry Residue Incineration in terms of "Climate Change"

3.1.3 Acidification/Eutrophication

The MSW management phase of putrescibles composting has an induced impact on the category Acidification/Eutrophication as well as contributes to the negative results of the damage macro-category Ecosystem Quality. A similar result was obtained by Eriksson et al. (2004) considering the installation of an anaerobic digestion plant. While, different results were obtained by Salhofer et al. (2007), who found lower impact in terms of the Eutrophication of mechanical biological treatment rather than incineration. The avoided impact is due to the energy recovery with the subsequent saving of fossil fuels. This amount decreases with the increasing of the percentage of separate collection up to 60%, while for greater percentages the maximum benefit is given by the glass production. The results are shown in Figures 8 and 9.

3.1.4 Carcinogens

In scenarios with the incineration of dry residue (1-10), the avoided impact increases with the percentage of separate collection due to the progressive reduction of the contribution of the incineration process. The main contribution in positive terms was given by the energy saving deriving from non-renewable sources.

In relation to scenario 20 (80% separate collection, mechanical sorting of dry residue), the direct landfill disposal of dry residue (scenario 21) produced an increase of about one order of magnitude in terms of the sub-category Carcinogens, thus determining a negative result in terms of the damage category Human Health. Similarly, the landfilling of inert materials and ashes of the combustion process (scenario 20) resulted in a negligible impact than that due to the direct landfilling of dry residue in scenario 21. Similar results were obtained by Bruno et al. (2002) who showed a significant impact of landfilling due to the release of heavy metals downstream leachate treatment.

Figures 10 and 11 show the trend of induced impacts in terms of the damage category "Carcinogens" by incineration and inert waste landfilling disposal, respectively.

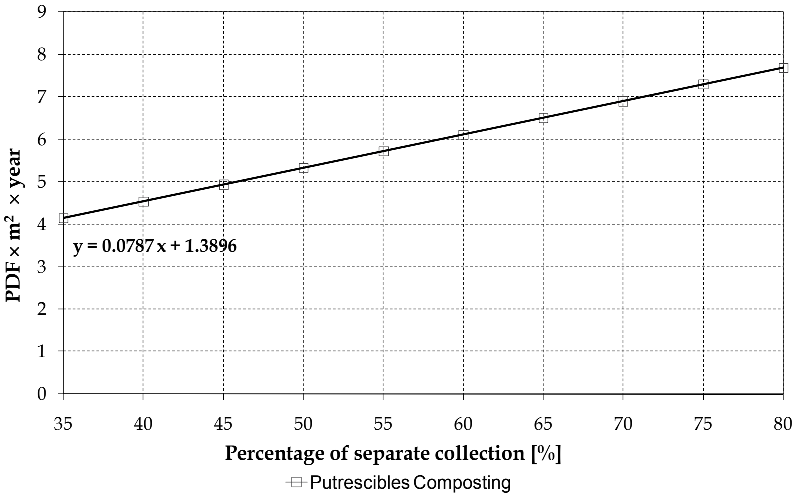

Fig. 8. Trend of induced impacts by the putrescibles composting in terms of damage category "Acidification/Eutrophication"

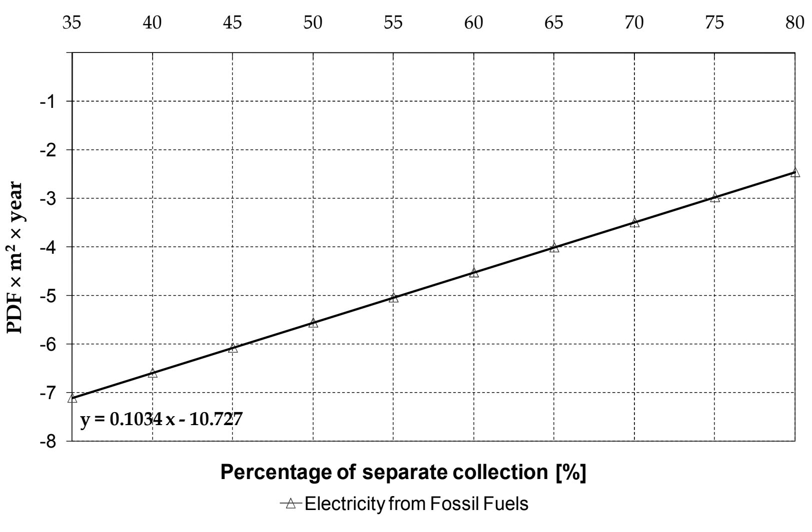

Fig. 9. Trend of induced impacts by the fossil fuels consumption in terms of damage category "Acidification/Eutrophication"

3.2 Comparison of the results obtained with WISARD and SimaPro

One of the aims of this study was to compare the results obtained with the application of two LCA procedures, WISARD and SimaPro, the first specific to the waste sector, while the second of a general nature. In particular, applicability and reliability of the single procedure to assess the life cycle of MSW management systems was evaluated.

It can therefore be deduced from the presentation of the results in the previous paragraphs that the comparison between the two procedures can be performed only in qualitative rather than quantitative terms because the mathematical models used for the analysis development as well as representation of the obtained data are completely different.

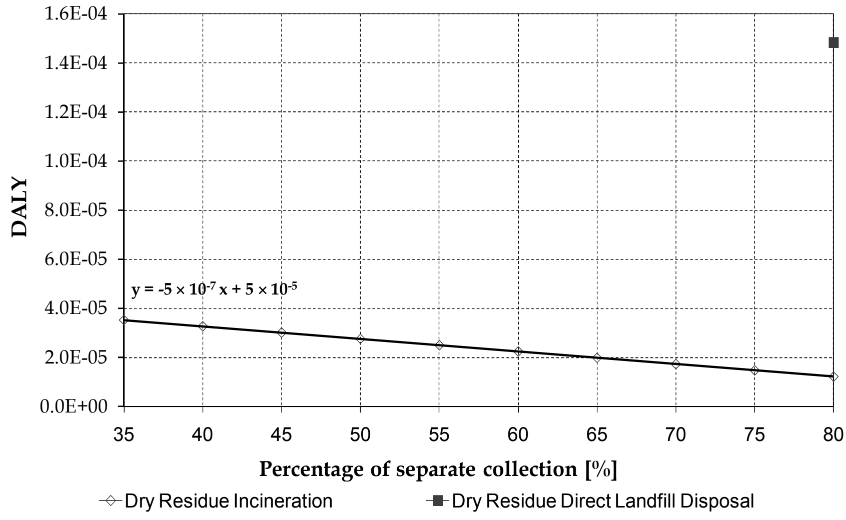

Fig. 10. Trend of induced impacts by incineration in terms of damage category "Carcinogens"

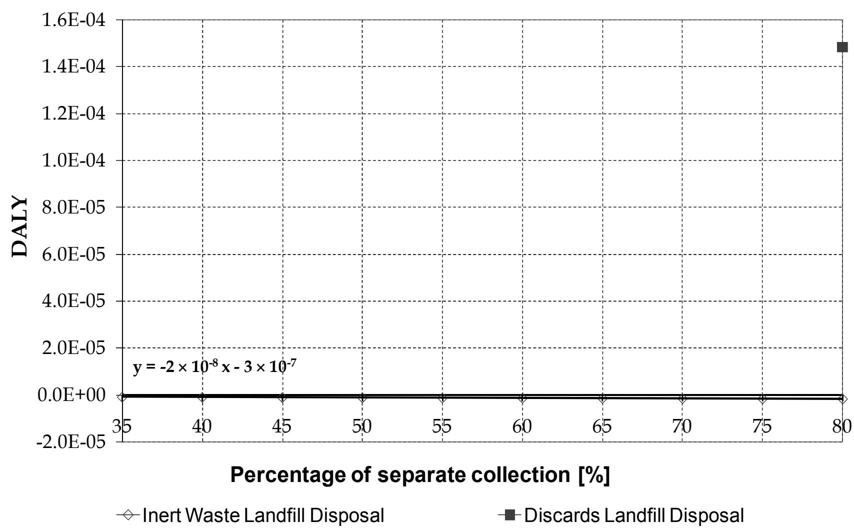

Fig. 11. Trend of induced impacts by inert waste landfill disposal in terms of damage category "Carcinogens"

Since the procedure WISARD is applied only to MSW management systems, with this procedure the results are presented only in terms of equivalent inhabitants. On the contrary, the SimaPro procedure, being of a general nature, can be adopted for the application of Life Cycle Assessment to all products, processes and activities. SimaPro, compared to WISARD, allows for a simpler and direct interpretation of the results, even by non-technical users. This is achieved with the presentation of the results in terms of the damage macro-categories Human Health, Ecosystem Quality and Resource Consumption.

The comparison between the results obtained with the two LCA procedures show the following similarities:

- all the considered scenarios showed negative overall impact indicators, indicating that the MSW management was environmentally sound compared with traditional methods of production of matter and energy. In particular, this behaviour was more evident the higher the percentage of waste collection;
- environmental emissions due to secondary production processes were lower than the corresponding emissions due to the primary production of packaging materials, with the presented exception;
- for a fixed percentage of separate collection, the solution with mechanical-biological selection of dry residue showed a reduction of the environmental benefit depending on the impact category take into account;
- for the percentages of separate collection greater than 60%, the solution with mechanical-biological selection of dry residue waste can be considered environmentally equivalent to the solution with the incineration and landfilling of ashes.

Table 19 shows the comparison between the MSW management phases with major and minor impacts for the WISARD and SimaPro procedures for the common Impact Category. Obviously, the MSW management phase with the greatest avoided impact indicates an environmental benefit, while the MSW management phase with the greatest produced impact indicates any environmental damage.

The qualitative comparison shows the perfect coincidence between the overall performances in terms of positive/negative values. While, the two LCA procedures showed a different behaviour in terms of the identification of the MSW management phase which affected the most the final result in terms of positive or negative impacts. The different behaviour is due to the different assumptions and simplifications made during the construction of the system and, particularly, in the implementation phase of the process units of the treatment and disposal plants.

As shown in table 9, focusing only on the common impact categories, Plastics and Metals Recycling and Glass Recycling was the MSW phase with the greatest avoided impacts for WISARD and SimaPro, respectively.

As shown in table 19, the collection and transporting to the treatment plants has a significant importance in the WISARD procedure, resulting as the phase with the greatest incidence on the production of induced impacts. The same results were not achieved for the SimaPro procedure because its general nature determined a major approximation in the construction of the basic calculation model.

3.3 Comparison with SimaPro between scenarios with dry residue incineration or sorting

The main aim of this paragraph is to compare the induced or avoided impacts due to scenarios with dry residue incineration (1-10) and scenarios with dry residue sorting (11-20), using SimaPro as an LCA tool. Firstly, it focused on the numerical results obtained with SimaPro for MSW management scenarios 11-20 developed in terms of avoided or produced impact. As shown in table 20, only for the damage category Acidification/Eutrophication do the impact values increase with the separate collection percentage, thus indicating an environmental negative effect due to the separate collection. On the contrary, for ten out of the eleven impacts considered with SimaPro, the calculated value decreased with the percentage of separate collection, thus confirming the environmental convenience to push

Impact Category	WISARD			SimaPro		
	Positive/ negative values – Trend	MSW phase with the greatest		Positive/ negative values – Trend	MSW phase with the greatest	
		Avoided impact	Produced impact		Avoided impact	Produced impact
Non Renewable Consumption	Negative - Decreasing	Plastics and Metals Recycling	Dry Residue Collection	Negative - Decreasing	Glass Recycling	Glass Primary Production
Greenhouses Gas Production	Negative - Decreasing	Plastics and Metals Recycling	Dry Residue Collection	Negative - Decreasing	Glass Recycling	Glass Primary Production
Acidification	Negative - Decreasing	Plastics and Metals Recycling	Composting	Negative - Decreasing	Glass Recycling	Composting

Table 19. Comparison between the MSW management phases with major and minor impacts for the WISARD and SimaPro procedures for the common Impact Category

up toward the maximization of separate collection. Moreover, for seven impact categories (see Table 20 for more details), the impact values were positive therefore indicating that they were avoided impacts (the integrated MSW management was environmentally sound in terms of these damage categories). While, the damage category "Land Use" showed both positive (for low levels of separate collection) and negative values (for high levels of separate collection). Finally, the impact values for the three damage categories Carcinogens, Ecotoxicity and Acidification/Eutrophication were only positive thus indicating that they were induced impacts (the integrated MSW management was not environmentally sound in terms of these damage categories). Table 20 gives the equation of the line giving the values of avoided or induced impacts by MSW management scenarios for each damage category. While, Tables 21 and 22 indicate the management phase with the greatest produced and induced impact for each impact category as well as for MSW management scenarios 11-20 developed in the study performed with SimaPro, respectively.

Table 21 shows the management phases with the greatest produced impact for each impact category as well as for MSW management scenarios 11-20 developed in the study performed with SimaPro. "Glass (Green)" resulted the heaviest phase 33 times out of 110, corresponding to 30%. While, "Landfill Disposal" was the heaviest phase 17 times, corresponding to 15.5%. Finally, "Wastewater treatment", "Glass Recycling", "Electricity consumption (nuclear)", "Natural fertilizers" and "Titanium dioxide production" were heaviest at the same manner: 10 times, corresponding to 9.1%.

Table 22 shows the management phases with the greatest avoided impact for each impact category as well as for MSW management scenarios 11-20 developed in the study performed with SimaPro. "Glass (White)" resulted the lightest phase 70 times out of 110 (10 scenarios x 11 impact categories), corresponding to 63.7%. While, "Leachate disposal", "Radioactive emissions", "Softwood" and "Bauxite consumption" were lightest at the same manner: 10 times, corresponding to 9.1%.

From this point forward, the aim of the paragraph is to compare scenarios 1-10 with scenarios 11-20 in order to qualitatively evaluate the environmental role of the incineration in the considered model of MSW management system. First of all, the difference between the values of Tables 20 (dry residue sorting scenarios) and 16 (dry residue incineration scenarios) were calculated in order to evaluate which scenarios are more environmentally sound in terms of the considered damage categories. The obtained results are condensed in figures 14 and 15. Essentially, for 10 out of the 11 impact categories (all excluding

Impact category	MSW management scenario									
	35% (11)	40% (12)	45% (13)	50% (14)	55% (15)	60% (16)	65% (17)	70% (18)	75% (19)	80% (20)
Carcinogens (DALY)	4.1E-04	3.7E-04	3.2E-04	2.8E-04	2.4E-04	2.0E-04	1.5E-04	1.1E-04	6.9E-05	2.6E-05
(+), decreasing	Impact (Daly)= 9×10^{-6} × (percentage of separate collection) + 7×10^{-4}									
Resp. Organics (DALY)	-1.7E-07	-2.2E-07	-2.6E-07	-3.1E-07	-3.6E-07	-4.0E-07	-4.5E-07	-4.9E-07	-5.4E-07	-5.8E-07
(-), decreasing	Impact (Daly)= 9×10^{-9} × (percentage of separate collection) + 1×10^{-7}									
Resp. Inorganics (DALY)	-2.3E-04	-2.5E-04	-2.7E-04	-2.9E-04	-3.1E-04	-3.3E-04	-3.5E-04	-3.7E-04	-3.9E-04	-4.1E-04
(-), decreasing	Impact (Daly)= 4×10^{-6} × (percentage of separate collection) - 1×10^{-4}									
Climatic Change (DALY)	-1.2E-06	-7.9E-06	-1.5E-05	-2.1E-05	-2.8E-05	-3.4E-05	-4.1E-05	-4.8E-05	-5.4E-05	-6.1E-05
(-), decreasing	Impact (Daly)= -1×10^{-6} × (percentage of separate collection) + 5×10^{-5}									
Radiation (DALY)	-1.7E-06	-1.8E-06	-2.0E-06	-2.2E-06	-2.4E-06	-2.5E-06	-2.7E-06	-2.9E-06	-3.1E-06	-3.2E-06
(-), decreasing	Impact (Daly)= -4×10^{-8} × (percentage of separate collection) - 4×10^{-7}									
Ozone Layer (DALY)	-5.7E-08	-6.2E-08	-6.8E-08	-7.3E-08	-7.9E-08	-8.5E-08	-9.0E-08	-9.6E-08	-1.0E-07	-1.1E-07
(-), decreasing	Impact (Daly)= -1×10^{-9} × (percentage of separate collection) - 2×10^{-8}									
Ecotoxicity (PDF*m^2*yr)	1.1E+03	1.0E+03	9.2E+02	8.1E+02	7.1E+02	6.1E+02	5.1E+02	4.1E+02	3.1E+02	2.1E+02
(-), decreasing	Impact (PDF*m^2*yr)= -20.29 × (percentage of separate collection) + 1828.60									
Acidif/ Eutroph. (PDF*m^2*yr)	4.0E-02	1.1E-01	1.8E-01	2.4E-01	3.1E-01	3.8E-01	4.4E-01	5.1E-01	5.8E-01	6.5E-01
(+), increasing	Impact (PDF*m^2*yr)= 0.0135 × (percentage of separate collection) – 0.4314									
Land Use (PDF*m^2*yr)	1.8E-01	3.4E-02	-1.1E-01	-2.6E-01	-4.1E-01	-5.6E-01	-7.1E-01	-8.6E-01	-1.0E+00	-1.2E+00
(+, -), decreasing	Impact (PDF*m^2*yr)= -0.0297 × (percentage of separate collection) + 1.2205									
Minerals (MJsurplus)	-5.0E+01	-5.5E+01	-6.0E+01	-6.5E+01	-7.0E+01	-7.4E+01	-7.9E+01	-8.4E+01	-8.9E+01	-9.4E+01
(-), decreasing	Impact (MJsurplus)= -0.9666 × (percentage of separate collection) – 16.45									
Fossil Fuels (MJsurplus)	-2.8E+02	-3.0E+02	-3.3E+02	-3.6E+02	-3.9E+02	-4.2E+02	-4.5E+02	-4.8E+02	-5.1E+02	-5.4E+02
(-), decreasing	Impact (MJsurplus)= -5.9412 × (percentage of separate collection) -67.324									

Table 20. Summary of the numerical results obtained with SimaPro for MSW management scenarios 11-20 developed in terms of avoided or produced impact. (-) = avoided impact, (+) = induced impact. Decreasing = the avoided or induced impact decreases with the increasing of separate collection percentage; Increasing = the avoided or induced impact increases with the increasing of separate collection percentage

Impact category	MSW management scenario									
	35% (11)	40% (12)	45% (13)	50% (14)	55% (15)	60% (16)	65% (17)	70% (18)	75% (19)	80% (20)
Carcinogens	Wastewat. treatment	Wastewat. treatment	Wastewat. treatment	Wastewat. treatment	Wastewat. treatment	Wastewat. treatment	Wastewat. treatment	Wastewat. treatment	Wastewat. treatment	Wastewat. treatment
Resp. Organics	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)
Resp. Inorganics	Glass Recycling	Glass Recycling	Glass Recycling	Glass Recycling	Glass Recycling	Glass Recycling	Glass Recycling	Glass Recycling	Glass Recycling	Glass Recycling
Climatic Change	Landfill disposal	Landfill disposal	Landfill disposal	Landfill disposal	Landfill disposal	Landfill disposal	Landfill disposal	Glass (Green)	Glass (Green)	Glass (Green)
Radiation	Electricity consumpt. (nuclear)	Electricity consumpt. (nuclear)	Electricity consumpt. (nuclear)	Electricity consumpt. (nuclear)	Electricity consumpt. (nuclear)	Electricity consumpt. (nuclear)	Electricity consumpt. (nuclear)	Electricity consumpt. (nuclear)	Electricity consumpt. (nuclear)	Electricity consumpt. (nuclear)
Ozone Layer	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)
Ecotoxicity	Landfill disposal	Landfill disposal	Landfill disposal	Landfill disposal	Landfill disposal	Landfill disposal	Landfill disposal	Landfill disposal	Landfill disposal	Landfill disposal
Acidif/ Eutroph.	Compost.	Compost.	Compost.	Compost.	Compost.	Compost.	Compost.	Compost.	Compost.	Compost.
Land Use	Natural fertilizers	Natural fertilizers	Natural fertilizers	Natural fertilizers	Natural fertilizers	Natural fertilizers	Natural fertilizers	Natural fertilizers	Natural fertilizers	Natural fertilizers
Minerals	Titanium dioxide product.	Titanium dioxide product.	Titanium dioxide product.	Titanium dioxide product.	Titanium dioxide product.	Titanium dioxide product.	Titanium dioxide product.	Titanium dioxide product.	Titanium dioxide product.	Titanium dioxide product.
Fossil Fuels	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)	Glass (Green)

Table 21. Management phase with the greatest produced impact for each impact category and for MSW management scenarios 11-20 developed in the study performed with SimaPro

"Minerals"), the difference was positive therefore indicating that sorting scenarios were heavier than the corresponding incineration scenarios. In particular, Figure 14 shows the trend of the difference between the Sorting scenario impact and Incineration scenario impact normalized in respect to the maximum impact value of each category for the following damage categories: "Carcinogens", "Resp. Organics", "Resp. Inorganics", "Climatic

Change", "Radiation", "Ozone Layer", "Ecotoxicity", "Acidif/Eutroph.", "Land Use" and "Fossil Fuels". As clearly shown in Figure 12, in terms of one of the ten listed impact categories, an Incineration scenario is more environmentally sound than the corresponding Sorting scenario with the difference linearly decreasing with the increasing of the percentage of separate collection.

Impact category	MSW management scenario									
	35% (11)	40% (12)	45% (13)	50% (14)	55% (15)	60% (16)	65% (17)	70% (18)	75% (19)	80% (20)
Carcinogens	Leachate disposal	Leachate disposal	Leachate disposal	Leachate disposal	Leachate disposal	Leachate disposal	Leachate disposal	Leachate disposal	Leachate disposal	Leachate disposal
Resp. Organics	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)
Resp. Inorganics	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)
Climatic Change	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)
Radiation	Radioact. emissions	Radioact. emissions	Radioact. emissions	Radioact. emissions	Radioact. emissions	Radioact. emissions	Radioact. emissions	Radioact. emissions	Radioact. emissions	Radioact. emissions
Ozone Layer	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)
Ecotoxicity	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)
Acidif/ Eutroph.	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)
Land Use	Softwood	Softwood	Softwood	Softwood	Softwood	Softwood	Softwood	Softwood	Softwood	Softwood
Minerals	Bauxite consumpt.	Bauxite consumpt.	Bauxite consumpt.	Bauxite consumpt.	Bauxite consumpt.	Bauxite consumpt.	Bauxite consumpt.	Bauxite consumpt.	Bauxite consumpt.	Bauxite consumpt.
Fossil Fuels	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)	Glass (White)

Table 22. Management phase with the greatest avoided impact for each impact category and for MSW management scenarios 11-20 developed in the study performed with SimaPro

Only for the damage category "Minerals" the difference between the Sorting scenario impact and Incineration scenario impact was negative, thus indicating that sorting scenarios were lighter than the corresponding incineration scenarios. As clearly shown in Figure 13, a Sorting scenario is more environmentally sound than the corresponding Incineration scenario with the difference linearly decreasing with the increasing of the percentage of separate collection.

Since for ten out of the eleven impact categories, the difference between the impact of a Sorting scenario and the impact of the corresponding (in terms of percentage of separate collection) Incineration scenario was positive, it can be argued that in general Incineration scenarios are more environmentally sound than the corresponding Sorting scenarios, especially for low levels of separate collection. While, on the contrary, the difference tends to diminish with the increasing of the percentage of separate collection.

4. Conclusion

The outputs from 12 out of 21 options modelled were initially analysed under eleven environmental effect categories as suggested by the WISARD procedure, with the aim of carrying out a synthetic study of the data available. The impact assessment categories suggested are as follows: renewable energy use, non-renewable energy use, total energy use, water, suspended solids and oxydable matters index, mineral and quarried matters, greenhouse gases, acidification, eutrophication, hazardous waste, non-hazardous waste. Attention was given to both measuring the overall impact due to the application of the

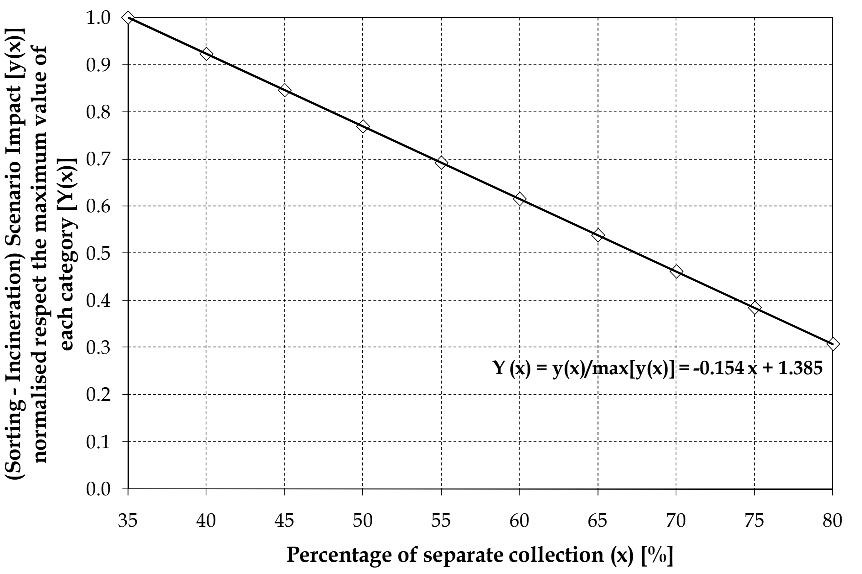

Fig. 12. Trend of the difference between the Sorting scenario impact and Incineration scenario impact normalized in respect to the maximum impact value of each category for the following damage categories: “Carcinogens”, “Resp. Organics”, “Resp. Inorganics”, “Climatic Change”, “Radiation”, “Ozone Layer”, “Ecotoxicity”, “Acidif/Eutroph.”, “Land Use” and “Fossil Fuels” (the positive value indicates that in terms of this impact category, an Incineration scenario is more environmentally sound than the corresponding Sorting scenario)

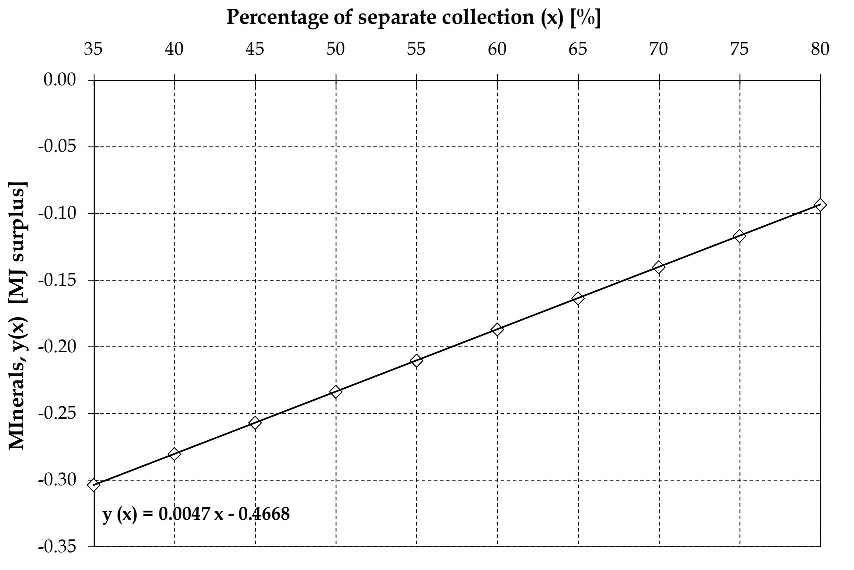

Fig. 13. Trend of the difference between the Sorting scenario impact and Incineration scenario impact for the damage category “Minerals” (the negative value indicates that in terms of this impact category a Recycling scenario is more environmentally sound than the corresponding Incineration scenario)

entire MSW management system adopted, as well as the evaluation of the specific contribution produced by each phase of the MSW management system.
The principal conclusions were that the scenario with 80% separate collection, no RDF incineration and dry residue sorting was the most environmentally sound option for six impact categories of the eleven chosen: renewable energy use, total energy use, water, suspended solids and oxydable matter index, eutrophication and hazardous waste. The second-best scenario with three impacts of environmentally sound categories (non-renewable energy use, greenhouse gases and acidification) is 80% separate collection, RDF production and incineration. For eight impact categories (renewable, non-renewable, total energy use, water, suspended solids and oxydable matter index, acidification, eutrophication, hazardous waste), all the MSW management scenarios produced a negative impact and the highest percentage of separate collection corresponded to the highest avoided impact.
A similar analysis was made with SimaPro considering the following impact assessment categories: Carcinogens, Respiration Organics, Respiration Inorganics, Climate Change, Radiation, Ozone Layer, Ecotoxicity Acidification/Eutrophication, Land Use, Minerals, Fossil Fuels. Analysing the emission data obtained on material from packaging shows that in most cases for each item impacts emissions of pollutants in secondary productions are lower than those corresponding of primary productions.
From a more detailed analysis of results, it appeared that all the scenarios considered have impact indicators relating to human health and resources with negative values. This means that in these cases, the integrated systems waste management are environmentally advantageous compared to traditional methods of production of matter and energy. In particular, the solution with the direct landfilling of residue is not preferred over the solution with waste incineration because there is more production of carcinogens substances during incineration and landfilling of ashes. The stages of management that most affect the final results of impact are incineration of waste, disposal in landfills, composting of organic and glass production.
Comparing the two calculation methods adopted, a coincidence of the results in terms of quality and performance is evident, highlighting the feasibility of the two procedures as well as the validity of the results. However, the same results are strongly influenced by the assumptions at the base of the building model and the approximations of reality, thus not making it possible to carry out a quantitative comparison due to the different models used for the characterization and representation of the results.
The results are similar for both Life Cycle Assessment procedures in qualitative terms. The study emphasized the priority of separate collection and recycling to save energy as well as reduce the environmental impact of MSW management.
The analysis carried out only with SimaPro, showed that for ten out of the eleven impact categories, the difference between the impact of a Sorting scenario and the impact of the corresponding (in terms of percentage of separate collection) Incineration scenario was positive, thus highlighting that in general Incineration scenarios are more environmentally sound than the corresponding Sorting scenarios, especially for low levels of separate collection. While, the difference tends to diminish with the increasing of the percentage of separate collection.

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24

Life Cycle Assessment in Municipal Solid Waste Management

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1. Introduction

Integrated Municipal Solid Waste (MSW) management is a tedious task requiring the simultaneous fulfilment of technical, economical and social constraints. It combines a range of collection and treatment methods to handle all materials in the waste stream in an environmentally effective, economically affordable and socially acceptable way (McDougall, 2001). Due to the complexity of the issues required for effective integrated MSW management, various computer-aided approaches that help the decision makers reach their final decision have been engaged since the early days of integrated MSW management. Any computer-based system supporting decision making is defined as a DSS (Finlay, 1989). DSS incorporate computer-based models of real life biophysical and economic systems. There are two main categories of DSS applied to solid waste management: the first one, based on applied mathematics, emphasises application of statistical, optimisation or simulation modelling. The second category of DSS provides specific problem-solving expertise stored as facts, rules and procedures. In addition, there are also hybrid approaches.

Recently, there has been a major shift towards Life Cycle Assessment (LCA) computer-aided tools. LCA is a holistic approach that is increasingly utilised for solid waste management especially in the decision-making process and in strategy-planning. LCA can be categorised as a hybrid approach since it utilises equations for inventory analysis and recycling loops on the one hand, while on the other it requires expertise input for impact assessment and characterisation.

Life Cycle Assessment (LCA) is a holistic approach that quantifies all environmental burdens and therefore all environmental impacts throughout the life cycle of products or processes (Rebitzer et al. 2004). LCA is not an exact scientific tool, but a science-based assessment methodology for the impacts of a product or system on the environment (Winkler & Bilitewski 2007). It is increasingly utilised for solid waste management systems especially in the decision-making process and in strategy-planning. LCA has been utilised for sustainable MSW management since 1995 (Güereca et al. 2006). LCA is an ideal tool for application in MSW management because geographic locations, characteristics of waste, energy sources, availability of some disposal options and size of markets for products derived from waste management differ widely (White et al., 1997; Mendes et al., 2004). LCA can help reduce local pressures and waste management costs, while considering the broader effects and trade-offs felt elsewhere across society (Koneczny and Pennington, 2007).

The LCA procedure has been standardized in 1998 and revised in 2006 (ISO 14040, 2006). Based on this standard, LCA consists of the following four sections:

- Goal and scope definition,
- Life cycle inventory (LCI),
- Life cycle impact assessment (LCIA),
- Life cycle interpretation.

2. Objective of the chapter

The objective of this chapter is the critical presentation of recent peer-reviewed research articles dealing with various stages of MSW management using the LCA methodology. In each article the main LCA components are presented (Goal and scope, functional unit, main assumptions, data sources for the compilation of the LCI, LCIA categories) in addition to the main conclusions of the study. Based on this review, conclusions are drawn for answering the key chapter question "What have we learned from the application of LCA to MSW?"

3. The challenge of dealing with the life cycle of MSW management

The application of LCA in MSW management is a very challenging task due to the following reasons:

- Every single waste management facility is considered *a priori* as environmentally friendly. However, solid waste management facilities require land (a lot of land in the case of landfills), consume non renewable natural resources for their operation (e.g. fuels and electricity) and emit a series of air pollutants and leachates. Therefore, waste management facilities put an environmental burden of their own on the natural environment. The trade-offs between environmental gains and burdens have to be assessed.
- Solid waste management facilities on the other hand generate a lot of useful "products"; Material reclamation facilities produce different sorts of paper and cardboard, glass, plastics, etc. A mechanical biological treatment facility generates RDF, which can be used as a solid fuel in cement kilns for example, and compost which can be used as a fertilizer substitute. Thermal treatment facilities, the so called waste-to-energy, produce electricity and heat. Therefore, solid waste management facilities have to be credited for all those useful "products".
- There is a great deal of uncertainty in a lot of the major solid waste treatment processes. The lack of quality data with respect to waste management practices is a recognized problem of LCA (McDougall, 2001). Landfilling, the most widely used MSW management option, has a lot of uncertainties related to the time frame of the impacts. Obersteiner et al. (2007) report that data relating to processes with direct measurements (such as collection, recycling and treatment) are more reliable than data from landfills which partially have to be modelled and where estimations are necessary.

4. The life cycle of MSW

The life cycle of MSW is depicted in Figure 1 by the dotted line. The LCA system boundary is the interface between the waste management system and the environment or other product systems. The life cycle starts once a material or product becomes waste, i.e. its owner discards it in the waste collection bins. MSW is collected either via mixed-bags or via separate collection. Each collection method requires its own infrastructure, i.e. dedicated bins and collection vehicles. The transportation stage follows. In the MSW management system of developed countries, the mixed bag waste can either go to the landfill, the waste-to-energy facility or to the Mechanical Biological treatment plant (MBT). The source-separated waste, if it is a dry stream (paper and cardboard, plastics, glass, tin, aluminium, etc.), can go to the material reclamation facility or if it is a wet stream (kitchen leftovers, garden trimmings, etc.) can go directly to the biological treatment facility.

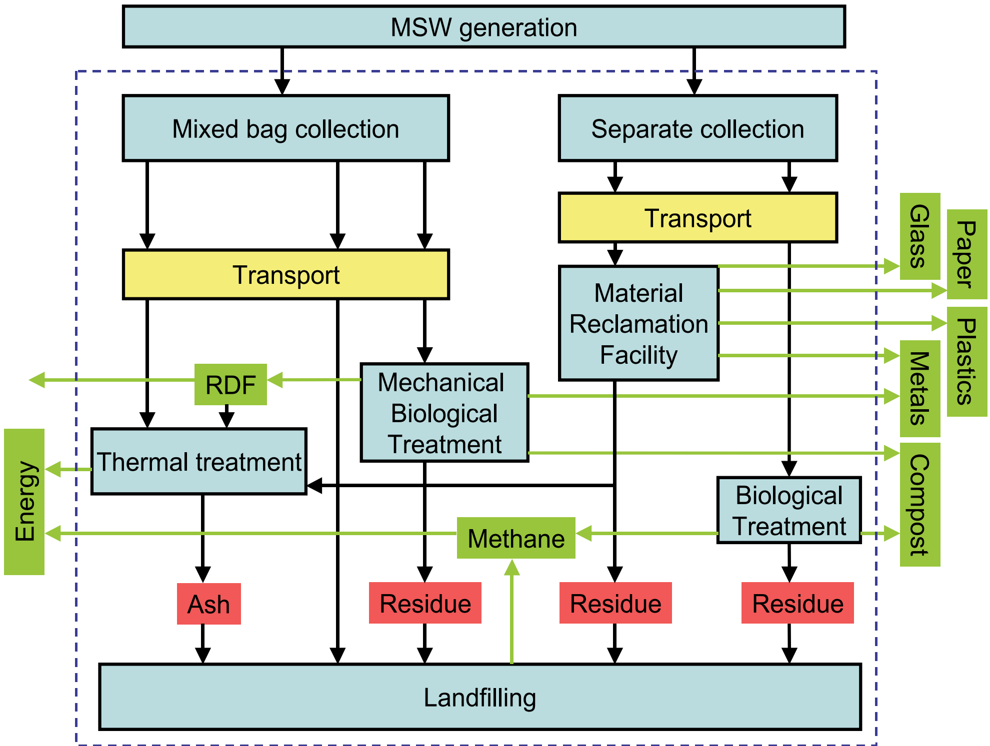

Fig. 1. The complete life cycle of MSW

In every management stage, products are presented in green boxes while residues from each management stage end up in the landfill. The end of the life cycle of MSW is when it ceases to be waste by becoming a useful product, residual landfill material or an emission to either air or water (McDougall, 2001). Landfill, therefore, is an end of the MSW life cycle. The production of useful products resulting from material or energy recovery is also an end of the life cycle of MSW. Figure 1 presents all possible routes for MSW management. This does not imply that each waste stream undergoes every management and treatment step. Please also note that Fig. 1 does not present the resources consumed and the emissions in each management step.

In the following paragraphs, each one of the aforementioned management stages is discussed and the necessary data for the implementation of their life cycle inventory are presented.

5. The life cycle inventory of MSW management

LCA assesses the use of resources and the release of emissions to air, water, land and the generation of useful products. All these inputs (material and energy resources) and outputs (emissions and products) have to be identified and quantified during the life cycle inventory (LCI) phase of the LCA. In the following sections, the most important LCI components of each management stage are identified and presented. Inputs from natural resources and output emissions are identified in red colour while the useful products in green. The functional unit (FU) is the reference to which the inputs and outputs are related (ISO 14040, 2006).

5.1 Collection and transport

Collection of MSW can either be in mixed bags or in separate bins. Mixed bag collection is the most widely applied method; however separate collection is a prerequisite for successful material recovery. Fig. 2 presents the inputs and outputs to the collection and transport stages of MSW management. The inputs are MSW and the materials and energy for the required infrastructure (MSW temporary storage containers and vehicles needed for collection and transportation). The outputs of these processes are again MSW (with altered physical properties such as density) and air, water and solid emissions.

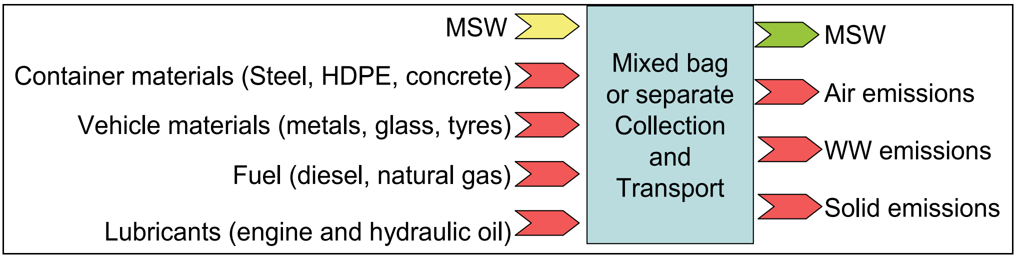

Fig. 2. Life cycle inventory components for the collection and transportation stages.

The following parameters must be taken into account for the compilation of an effective LCI in the collection and transportation stages of an LCA:

- Selective collection system,
- Material of containers (HDPE, steel and fiber glass),
- Collection frequency,
- Distance covered,
- Type of collection truck (pneumatic, top loader, rear loader, side loader),
- Fuel of collection truck (diesel, natural gas),
- Density of the waste fractions in containers and collection trucks,
- Size of containers,
- Filling percentage of the waste containers.

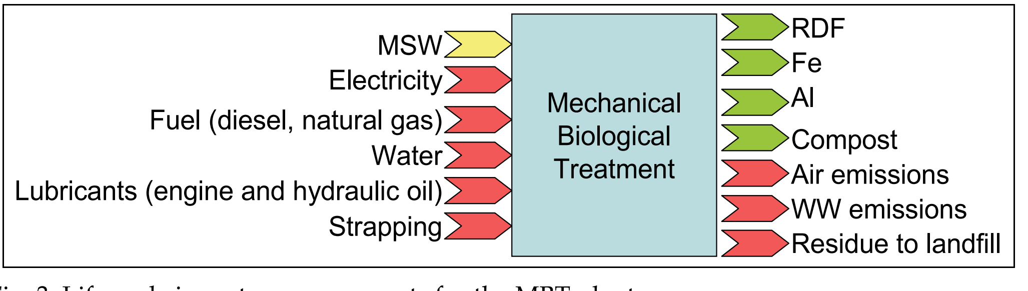

Fig. 3. Life cycle inventory components for the MBT plant.

5.2 Mechanical and biological treatment

Mechanical and biological treatment is a process that generates many useful "products" (see Fig. 3). Its inputs include mixed-bag MSW, electricity, fuels (e.g. diesel and natural gas), water and materials for the required infrastructure (e.g. lubricants and strapping). The outputs are recovered metals (Fe and Al), RDF (which ultimately can be used as an energy source), compost (which can substitute chemical fertilisers), emissions to air and water and finally a fraction of residue that ends up in the landfill.

5.3 Thermal treatment

The major inputs and outputs considered when compiling the LCI of an incineration plant are the following (see Fig. 4): MSW, electricity, other fuels (diesel, natural gas or even coal), water and activated carbon (for air pollution control), are the major inputs. On the other hand, the outputs are: flue gas (HCl, SO_2, NO_x, dioxins, CO, PM10, HF), bottom ash, iron scrap, electricity generated, water discharge and air pollution control residues.

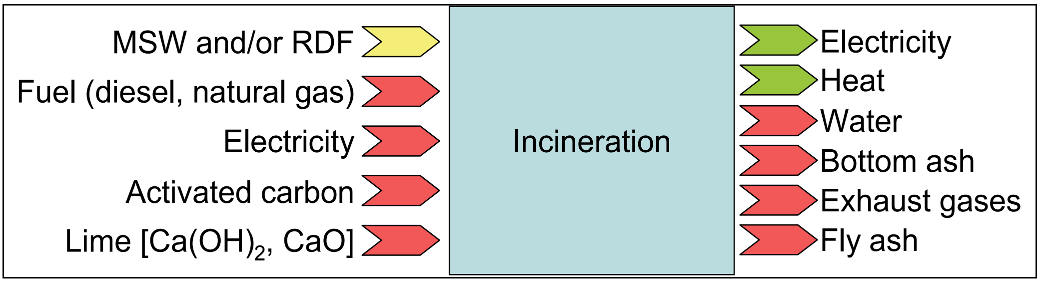

Fig. 4. Life cycle inventory components for an incineration plant.

The key factors in modelling incineration in LCA terms are (Chen & Christensen, 2010): incineration technology (e.g. grated firing, fluidized bed), the heating value of MSW (specified by the MSW composition), the use of auxiliary fuel (type and amount) and leachate disposal methods (e.g. spraying, wastewater treatment).

5.4 Biological treatment

Fig. 5 presents the major inputs and outputs for the life cycle inventory of MSW biological treatment. There are two processes included under the term "biological treatment" in MSW management: composting and anaerobic digestion. The biodegradable fraction of the MSW is involved in both of the aforementioned processes. Composting is an aerobic process. The degradable organic carbon in the MSW is converted into CO_2.

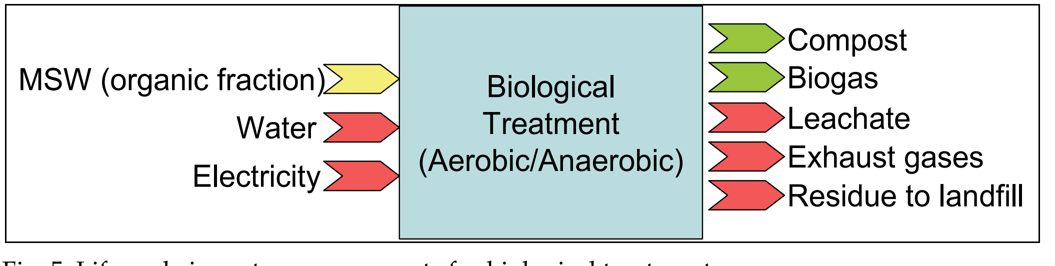

Fig. 5. Life cycle inventory components for biological treatment.

5.5 Landfilling

Landfilling is the first and oldest MSW treatment option. The types of landfilling facilities, all over the world, range from uncontrolled dumpsites to highly engineered facilities with leachate and lanfill gas (LFG) management. Fig. 6 presents the major inputs and outputs for the life cycle inventory of landfilling. When MSW is landfilled directly, anaerobic biological degradation produces landfill gas and leachate. Over 90% of the converted organic carbon is released as CO_2 and CH_4. The remainder is released in the leachate (Obersteiner et al., 2007).

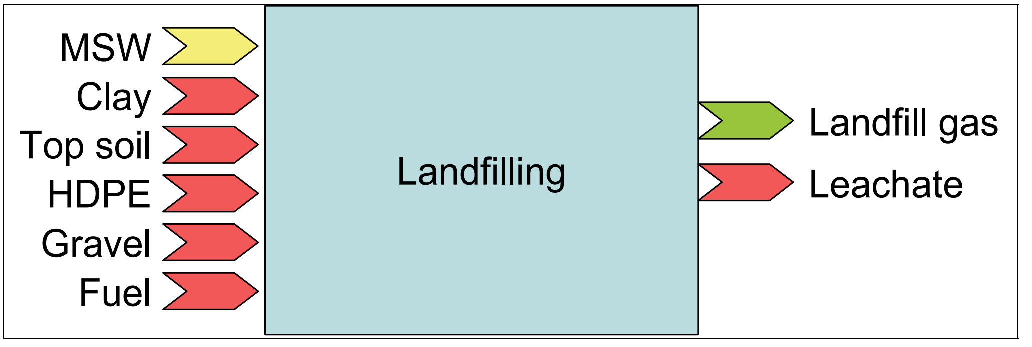

Fig. 6. Life cycle inventory components for landfilling.

Environmental impacts arising from landfills are: leachate (heavy metals and organic loading), emissions into the air (CH_4, other hydrocarbons), the energy inputs (fuel and electricity) and material inputs for the construction of the engineered landfills (HDPE, clay, gravel, top soil).

5.6 The contribution of capital equipment and infrastructure

Waste management systems require capital equipment and infrastructure for their operation, in addition to inputs of energy and materials. All of these equipment and infrastructure consume natural resources and release emissions to the environment during their respective life cycles. These emissions, also known as secondary environmental burdens, tend to be excluded from LCAs of MSW since they are assumed to be relatively small in comparison to primary burdens (McDougall et al., 2001).

6. Review of selected peer reviewed publications

All of the reviewed studies appeared recently in peer-reviewed journals. They are presented in chronological order starting from the oldest. They are comparative LCAs that evaluate the consumption of natural resources, environmental emissions and/or performance of various types of MSW management systems. The MSW management stages considered in the reviewed publications are the following:

- Collection and transport,
- Material recovery via separate collection, material recovery facilities or the application of MBT technology,
- Thermal (mostly incineration) and biological treatment (both composting and anaerobic digestion) treatment,
- Final disposal via landfilling.

Mendes et al. (2003) examine the management of the biodegradable MSW fraction in Sao Paulo, Brazil.
Goal and scope: The goal of the study was to compare composting, biogasification and landfilling. The scope included the analysis of 5 scenarios: i) landfilling, ii) landfilling with energy recovery, iii) composting, iv) composting followed by gas treatment (compost with biofilter) and v) biogasification.
Functional unit: the treatment and disposal of 1 ton of MSW
LCI: the main sources of data were published Japanese LCA reports
Software used: None
Assumptions: Emissions from the construction of facilities were not included in the study because they are assumed to be small compared to those released during the operational stage of the facilities.
LCIA: based on 3 impact categories: global warming potential, acidification potential and nutrient enrichment potential.
Main conclusions: Landfilling was the scenario with the highest environmental impacts, except in the case of acidification potential, in which composting presented the highest potential. Composting without gas treatment presented higher environmental impacts than biogasification. Finally, both composting and biogasification can decrease significantly the impacts compared to landfilling. The authors also mention that both waste composition and carbon intensity of energy sources are very important factors to the outcome of the environmental impact of an MSW management system.

Beigl & Salhofer (2004) compare different waste management systems of rural communities in the region of Salzburg in Austria.
Goal and scope: The goal of the study was to compare the ecological effects and costs of different waste management systems in a selected rural area in Austria. The scope of the study included 3 scenarios: scenario 1 included recycling by collection in the bring system; scenario 2 included recycling by kerbside collection; scenario 3 was non-recycling.
Functional unit: the amount of communal waste generated annually
LCI: data from the actual practices of collection and treatment were used.
Software used: IWM
Assumptions: Switzerland in 1997 was chosen as the area and year of reference for comparison purposes due to the lack of Austrian data
LCIA: The impact categories examined were the global warming potential, the acidification potential and the net energy use. However, no life cycle impact assessment phase was included, therefore the study is not really an LCA.
Main conclusions: Kerbside collection is ecologically better than collection in the bring system because the specific fuel consumption is lower for collection transports than that for individual transports. With regard to acidification and net energy use, the recycling of metals plays an important role.

Hischier et al. (2005) study the application of LCA on the management of a certain fraction of MSW, namely the waste of electrical and electronic equipment (WEEE).
Goal and scope: The examination in environmental terms of the two Swiss take-back and recycling systems of SWICO (for computers, consumer electronics and telecommunication equipment) and S.EN.S (household appliances).

Functional unit: All activities linked with the disposal and recycling of WEEE accumulated over one year (2004) in Switzerland.
LCI: Data are derived from the two separate WEEE recycling systems that operate in Switzerland: the SWICO Recycling Guarantee and the S.EN.S system. Each of these systems covers different parts of WEEE. The two systems are well established in Switzerland; In 2004 the systems yielded the recycling of 11 kg of WEEE per inhabitant, a figure well over the goal of 4 kg of WEEE recycled defined in the European WEEE directive.
Software used: None
LCIA: Based on the impact categories from the CML methodology were used.
Main conclusions: The take-back and recycling system for WEEE as established in Switzerland has clear environmental advantages, compared to the complete incineration of all WEEE.

Hong et al. (2006) apply LCA to study MBT application in China.
Goal and scope: Comparison of the environmental impact potential of five different alternative waste treatment strategies: i) landfill, ii) incineration, iii) Biological and mechanical treatment (BMT)-compost, iv) BMT-incineration and v) BMT-landfill. In scenario 3, MSW is firstly pre-treated by BMT and then be composted.
Functional unit: Treatment of 2200 t/day of MSW in the Pudong New Area, in Shanghai, China.
LCI: The primary data come from the incineration plant, the biological compost plant, the landfill yard and Pudong Environmental Protection Bureau.
Software used: none
LCIA: Based on three impact categories: global warming potential (GWP), acidification potential (AP) and eutrophication potential (EP).
Main conclusions: The results of LCA show that the incineration process of MSW presents the highest acidification potential while the landfill presents both the highest global warming and eutrophication potential.

Özeler et al. (2006) study various MSW management methods for Ankara, Turkey.
Goal and scope: The goal of the study was the comparison among five scenarios that included different municipal solid waste processing and/or disposal methods. The management system components considered in the scenarios were: collection and transportation of MSWs, source reduction, material recovery facility/transfer stations, incineration, anaerobic digestion, and landfilling.
Functional unit: The amount of municipal solid waste generated in the districts of Ankara.
LCI: The data collection and preparation were mainly based on information provided by the Solid Waste Management System of Ankara.
Software used: IWM-1
LCIA: The IWM-1 model is an LCI model; therefore there is no explicit LCIA phase
Main conclusions: The scenario which included source reduction, collection, transport and landfilling was the one with minimum contribution in all the impact categories but global warming and FSW, due to the source reduction process and subsequent recycling of the sorted materials in addition to less solid waste input to landfill.

Wanichpongpan & Gheewala (2007) examine the landfill gas-to-energy conversion in Thailand.

Goal and scope: The goal of the study was to evaluate the reduction potential of methane gas emissions from MSW landfill. The scope of the study included two scenarios: Scenario 1 included a single landfill using the methane emitted for electricity production. Scenario 2 included two small landfills without electricity production and with flaring of the collected methane
Functional unit: 1 ton of collected MSW
LCI: data from municipalities were collected for the MSW collection and transportation. The Landfill Gas Emissions Model (LandGEM) was used for the quantification of air emissions from landfills. The UNFCCC guidelines were also used.
Software used: None
Assumptions: Leachate treatment is not included as it is common to both scenarios. Emissions from the construction of facilities are also not included since they are assumed small compared to those of the operating facilities.
LCIA: the only impact category of interest to the authors was the global warming potential.
Main conclusions: centralized landfills are viable with landfill gas-to-energy projects and preferable over the current management system of small landfills.

Chaya & Gweewala (2007) examine the MSW-to-energy schemes in Thailand.
Goal and scope: The goal was to compare the performance of two MSW-to energy schemes, incineration and anaerobic digestion, in terms of environmental impacts and energy balance.
Functional unit: 1 ton of MSW managed
LCI: data for incineration were obtained from a plant in Phuket in South Thailand. For anaerobic digestion, data were obtained from technical manuals and refereed literature.
Software used: SimaPro 5
Assumptions: transportation, construction and maintenance of the plants, and recycling were not included in the study.
LCIA: Based on the Ecoindicator 95 ready-made method
Main conclusions: MSW anaerobic digestion was preferable to incineration. This was partly because more than 60% of the waste is biodegradable and thus suitable for anaerobic digestion.

Buttol et al. (2007) examine the MSW management system of the Bologna district in Italy.
Goal and scope: The scope of the study was to compare different MSW management options in the Bologna district. The scope of the study included 3 different scenarios: scenario 1 is based on the current MSW practices; scenario 2 anticipates a strong increase in the fraction sent to incineration with energy recovery, the percentage increasing from 30% to 50% of the total MSW; scenario 3 anticipates a fraction sent to incineration equal to 37% of the total waste and a separated collection equal to 31%.
Functional unit: The collection and treatment of 566,000 tons of MSW, which correspond to the annual generation in the district of Bologna for 2006
LCI: Data were obtained from the actual MSW management operations in Bologna
Software used: WISARD
Assumptions: Are made on every management step, i.e. incineration with energy recovery, landfilling with energy recovery, composting, sorting and recycling.
LCIA: Based on the following impact categories: greenhouse effect, air acidification, eutrophication, depletion of non-renewable resources, ecotoxicity (sediment, terrestrial, aquatic), human toxicity.

Main conclusions: There is a clear environmental benefit in increasing recycling and incineration with energy recovery.

Liamsanguan & Gheewala (2008) examined two methods of MSW for the island of Phuket, Thailand.
Goal and scope: the goal of the study was the comparison of 2 waste management methods used currently for MSW management in the island of Phuket, namely landfilling (without energy recovery), and incineration (with energy recovery). The scope of the study was the comparison in terms of energy consumption and greenhouse gas emissions.
Functional unit: 1 ton of MSW treated
LCI: Information about energy consumption of the MSW management systems was collected from the actual processes at the study site. Emission factors used were based on refereed literature and commercially available databases (BUWAL 300, ETH-ESU).
Software used: None
Assumptions: The treatment of landfill leachate was not included in the study because its energy and resource requirements are negligible. Transportation of MSW was also not included in the study because it is common to both MSW management systems.
LCIA: this study is based just on the life cycle inventory, therefore it is not really an LCA
Main conclusions: Incineration was found to be superior to the landfilling. However, landfilling reversed to be superior when landfill gas is recovered for electricity production.

Iriarte et al. (2009) applied LCA to compare systems or subsystems of waste management and treatment and to identify which areas require an improvement in terms of environmental performance.
Goal and scope: The main objective of the study was to compare the overall environmental impacts of three selective collection services of MSW in dense urban area: i) mobile pneumatic, ii) multi-container, and iii) door to door systems.
Functional unit: The provision of the selective collection service of 1500 tons a month of MSW generated in an urban locality with a density of 5000 inhabitants/km^2, in a European setting, considering a rate of theoretical recovery of 100% for the following fractions: organic, paper, packaging and glass by means of the aforementioned three selective collection systems.
LCI: The data of the operations and infrastructure of the selective collection systems have been obtained from the field work of the members of the group, management reports and waste management programmes, container companies, waste collection truck suppliers and suppliers of pneumatic waste collection systems.
Assumptions: The main assumptions of the study refer to the fraction densities, the equipment and infrastructure, consumption of resources in waste transport and differences in the values of impact categories.
Software used: SimaPro 7.0.2
LCIA: Based on the CML 2 baseline 2000 method.
Main conclusions: The collection system with the least impact is multi-container collection. The mobile pneumatic system has the greatest environmental impact in the categories of global warming, fresh water aquatic ecotoxicity, terrestrial ecotoxicity, acidification and eutrophication. The door-to-door system has a greater environmental impact in the categories of abiotic depletion, ozone layer depletion and human toxicity. In addition, the door-to-door system has the highest energy demand. This result is mainly due mainly to the

waste urban transport associated to its longer collection routes. However, the authors claim that the low environmental performance of the door-to-door collection system should be seen in a wider context, since it delivers higher recovery rates of waste compared to the other collection options.

Cherubini et al. (2009) compare selected waste disposal alternatives in a life cycle perspective, considering both landfill systems, where no recycling takes place, and systems which are able to minimize the amount of landfilled waste while maximizing material and energy recovery.
Goal and scope: The goal of this study is to provide a transparent and comprehensive environmental evaluation of a range of waste management strategies for dealing with mixed waste fractions in the city of Roma, Italy. Regarding the scope of the assessment, four different waste management strategies are investigated: Scenario 0: wastes are delivered to landfill without any further treatment; Scenario 1: part of the biogas naturally released by the landfill is collected, treated and burnt to produce electricity; Scenario 2: a sorting plant is present at landfill site for separation of the organic and inorganic fractions and for ferrous metal recovery. Electricity, biogas and compost are then produced on site; Scenario 3: wastes are directly incinerated to produce electricity.
Functional unit: The amount of waste produced in a year (2003) by the city of Roma, which must be disposed of: 1460 kton of wastes contained in the so-called "black sacks" (i.e. pre-sorted and recycled wastes not included).
LCI: Data were compiled from selected references.
Software used: SimaPro 7.1
LCIA: based on global warming potential, acidification potential and eutrophication potential.
Main conclusions: Results show landfill systems (scenarios 0 and 1) are the worst waste management options and that significant environmental savings are achieved from undertaking energy recycling.

De Feo & Malvano (2009) study various MSW management scenarios in Southern Italy.
Goal and scope: The aim of this study was to apply the LCA procedure to MSW management on the Province of Avellino in Italy in order to choose the "best" management system in environmental terms. The MSW management scenarios considered can be divided into two categories: the first includes scenarios that are based on the incineration of the dry residue, while the second does not consider the thermal treatment of dry residue.
Functional unit:
LCI: All the data necessary for the construction of the analysed scenarios were deduced from the Province of Avellino and the two MSW management companies.
Software used: WISARD
Assumptions: The facility for the production of the RDF was simulated as an MBT plant.
LCIA: The 11 impact assessment categories applied are: renewable energy use, non-renewable energy use, total energy use, water, suspended solids and oxydable matters index, mineral and quarried, greenhouse gases, acidification, eutrophication, hazardous waste, non hazardous waste.
Main conclusions: The selection of the best scenario depends on the impact category examined. More specifically the scenario that includes 80% separate collection, no RDF incineration and dry residue sorting was the most preferable for the following six impact

categories: renewable energy use, total energy use, water, suspended solids and oxydable matters index, eutrophication and hazardous waste. On the other hand, the scenario with 80% separate collection and RDF production and incineration is preferable for the following three impact categories: non-renewable energy use, greenhouse gases and acidification. Finally, the scenario with 35% separate collection, RDF production and incineration is the most preferable for the mineral and quarried matters and non-hazardous waste impact categories.

Banar et al. (2009) study various MSW management methods for Eskisehir, Turkey.
Goal and scope: The goal of the study was to analyse and evaluate different alternatives that can be implemented to enable the targets required by the European Landfill and packaging and Packaging Waste Directives for solid waste management in the city of Eskisehir, Turkey. The scope of the study included the development of five alternative scenarios to the current MSW management system, which is uncontrolled dumping. Scenario 1 is an improved version of the current system assuming a 92.7% landfilling; Scenario 2: A source separation system with efficiency 50% was added as an improvement to scenario 1. The recyclables obtained from source separation were sent to the MRF; Scenario 3: The flow of recyclables is similar to scenario 2, while the organic fraction from the MRF is transported to the composting facility. Scenario 4: An incineration process was added instead of a composting facility. All organic wastes and the wastes from the separated recyclables are transported to the incinerator (85%); Scenario 5: all MSW is sent to the incineration facility (100%).
Functional unit: The management of 1 ton of MSW of Eskisehir.
LCI: Data were gathered from actual applications in Eskisehir, literature and the database of SimaPro 7.
Software used: SimaPro 7
LCIA: Based on 6 impact categories included by the CML method, namely: abiotic depletion, global warming, human toxicity, acidification, eutrophication, and photochemical oxidation.
Main conclusions: Recycling of materials leads to lower abiotic depletion. Also, the scenarios that include recycling (S2, S3 and S4) are better than the others in terms of human toxicity (mainly due to the recycling of aluminium). Scenario 3 is the best option in terms of global warming, acidification (because of the displacement of fertiliser), eutrophication and photochemical ozone depletion.

Khoo (2009) compares various waste conversion technologies in Singapore.
Goal and scope: The goal of the study is to compare various waste conversion technologies in Singapore. The scope of the study includes a total of eight waste treatment options for converting an assortment of waste types, including MSW, scrap wood and tyres, organic wastes and RDF into synthetic gas or product gas. All of the technologies are based on pyrolysis and gasification.
Functional unit: 1 ton of product gas produced from the assortment of waste materials
LCI: Data for the 8 technologies are compiled from various reports
Software used: None
LCIA: based on the EDIP 2003 methodology, the following impact categories are reported: global warming potential, acidification potential, terrestrial eutrophication and ozone photochemical formation.

Main conclusions: Pyrolysis-gasification of MSW and the steam gasification of wood are the most favourable candidates in terms of environmental performance.

Wittmaier et al. (2009) apply LCA in waste utilization systems in an unnamed region in Northern Germany
Goal and scope: The goal of the study was the assessment of the thermal treatment of waste in respect to climate change for various waste treatment systems. The scope included 2 scenarios. Scenario 1 was a conventional thermal treatment, i.e. a waste incineration plant with stroker-fired furnace and multistage flue gas cleaning. Scenario 2 was termed as optimized energy recovery and included the specific preliminary separation of the waste materials through mechanical treatment, followed by a subsequent conventional thermal treatment of the separated lower calorific waste fraction as described in Scenario 1. In both scenarios, the landfilling of combustion residues was defined as a further element of the system.
Functional unit: The treatment of 198,000 tonnes of MSW which correspond to the annual amount generated in the district
LCI: Data were compiled from literature and actual operations in Germany
Software used: GaBi 4
LCIA: The only impact category studied was the global warming potential
Main conclusions: The analyses presented in this study show that the thermal treatment of waste in a waste incineration plant can reduce emissions of greenhouse gases compared with depositing the same amount in a landfill, by half. Moreover, a further reduction of the greenhouse gases emissions can be achieved by the energetic utilization of waste with increased calorific value, which could not otherwise be advantageously used in a waste incineration plant.

Rives et al. (2010) compare container systems in MSW. The authors state that the selection of a particular type of waste container by an institution corresponds, in the majority of the cases, to economic or aesthetic criteria, but never to environmental ones. Therefore, the aim of their study is to analyse the potential environmental impact of fourteen MSW container systems, using LCA. The difference among the systems lies in the individual characteristics of the containers, especially the volume and weight of the manufactured materials.
Goal and scope: The objective as to compare and quantify the environmental impact of different MSW waste collection containers, based on their volume and manufacturing material.
Functional unit: The storage of collected and unsorted municipal solid waste (MSW) during the day, in an average neighbourhood of 1000 inhabitants, with a Spanish average waste generation of 1.47 kg/inhabitant/day and a density of waste container of 106 kg/m^3.
LCI: Nine HDPE and five steel containers were studied, ranging in volumes of 60 l to 2400 l.
Assumptions: MSW containers were completely full, containing identical composition of MSW, ii) unsorted waste collection was carried out on a daily basis, and iii) all waste generated was collected unsorted.
Software used: SimaPro 7.1
LCIA: Based on the CML 2 baseline 2000 method. The impact categories considered are: Abiotic depletion potential (ADP), Global warming potential (GWP), Ozone layer depletion potential (OLDP), Acidification potential (AP), Eutrophication potential (EP), Photochemical oxidation potential (POP), Human toxicity potential (HTP), Terrestrial ecotoxicity potential (TEP)

Main conclusions: A steady reduction in materials was observed as the volume of the waste container increases, for both the HDPE and steel containers. More specifically, the analysis showed that in order to satisfy the functional unit, the smaller volume HDPE container systems (60 l and 80 l) had the greatest environmental impact. This was true for the majority of the impact categories, except for the EP and HTP categories in which the 660 l and 770 l steel containers had the greatest impact.
A comparison of MSW containers of the same volume and different materials was carried out too. It was observed that HDPE container systems have 1.5-9 times greater environmental impact than the steel containers in most of the category impacts except in the EP, POP and HTP categories. Collection systems that use 2400 l steel waste containers have the least environmental impact.
Finally, sensitivity analysis showed that there is a direct dependence among the filling percentage of waste container, the waste collection frequency, the waste generation per capita and the density of the waste container's contents.

Chen & Christensen (2010) assessed the environmental profile of two MSW incineration technologies that are commonly used in China.
Goal and scope: The goal of the study is the comparison between two incineration technologies with semi-dry flue gas cleaning for treating MSW in southern China, namely grated firing and fluidized bed. The scope of the study included nine different scenarios based on the aforementioned incineration technologies.
Functional unit: 1 ton of waste arrived at the incineration plant
LCI: based on the databases of the software used
Software used: EASEWASTE
LCIA: Based on the EDIP 1997 method. The important impact categories related to incineration are: global warming (100 years), acidification, nutrient enrichment, human toxicity via soil, water and air, ecotoxicity, bulky waste, photochemical ozone formation, slag and ashes.
Main conclusions: for MSW with Lower Heating Value high enough for self-maintained combustion (e.g. as high as 6.05 MJ/kg) the fluidized bed incineration without coal consumption saves more potential impacts than grate furnace incineration technology for most of the evaluated impact categories.

Abduli et al. (2010) compare 2 different MSW management scenarios in Tehran, Iran.
Goal and scope: The goal of the study was to compare the environmental impacts of two MSW management practices. The scope was to compare landfill (scenario 1) and composting plus landfill (scenario 2) for the management of MSW in the city of Tehran.
Functional unit: 1 ton of MSW
LCI: Data gathered from actual applications in Tehran, literature and the database of LandGem model are used
Software used: None
Assumptions: Landfill has a gas collection system with 50% collection efficiency
LCIA: Seven impact categories are considered to be representative of the potential environmental impact of MSW management in Tehran: climate change, acidification, respiratory effect, carcinogenesis, ecotoxicity, ozone layer depletion and surplus energy for future extraction.

Main conclusions: The study shows that scenario 2 (composting plus landfill) has a higher environmental impact compared to landfilling, despite the fact that the fraction of organic waste in MSW is quite high (67.8%)

Miliūtė & Staniškis (2010) apply LCA on the MSW management systems in Lithuania.
Goal and scope: The goal of the study was to compare different waste management options for the MSW in the region of Alytus, Lithuania. The scope of the study included 5 different scenarios: Scenario 1 was based on landfilling; scenario 2 included recycling, composting and landfilling; scenario 3 was based on recycling, composting, MBT and incineration; scenario 4 was based on recycling and incineration while scenario 5 included recycling, MBT and incineration.
Functional unit: the MSW generated in one year (2005): 45,150 tonnes
LCI: waste composition data were extracted from empirical studies in the region of Alytus. Data were also extrapolated from official Lithuanian statistics. The data on incineration processes are based on the average Swedish technologies.
Software used: WAMPS
Assumptions: The time boundary of the study was set at 10 years. Assumptions are made for all the waste management options (incineration, landfilling, composting, recycling) of the study.
LCIA: based on 4 impact categories: global warming, acidification, eutrophication and photo-oxidant formation
Main conclusions: Landfilling gives the worst environmental results compared to the other waste management options. Furthermore, when it comes to the biodegradable waste fraction, aerobic composting is not a better option compared to incineration with energy recovery in all impact categories.

Morris (2010) compares waste-to-energy (WTE) and landfill (LF) gas for electricity generation in North America in terms of greenhouse gases (GHG) emissions.
Goal and scope: there are two goals in the study: the first one is to compare WTE and LF in terms of their climate impact; the second one is to compare MSW, natural gas and coal for power production in terms of climate impact.
Functional unit: for the comparison of Waste-to-energy and landfilling the FU is 1 metric ton of MSW shipped from a transfer facility to LF or WTE for disposal; for the comparison of the GHG releases for power production from MSW, natural gas and coal the FU is the amount of fuel required to produce 1 kilowatt hour (kWh)
LCI: data are based on three different levels of North American geographic specificity: the city of Seattle, the metropolitan area of Vancouver and the state of Massachusetts.
Software used: None
Assumptions: GHG emissions from construction of capital and operating equipment are not included in either inventory.
LCIA: the only impact category considered is climate change
Main conclusions: The author defines the “crossover rate” as the LFG capture rate at which burning and burying have equal GHG emissions. Above the crossover rate, LF has lower GHG emissions than WTE. Below the crossover rate, WTE is better for the climate. Seattle and Massachusetts crossover rates are higher than Metropolitan Vancouver, mainly due to to Seattle and Massachusetts MSW having lower fossil carbon content, which results in lower WTE fossil CO_2 emissions. Regarding the comparison for power generation, natural

gas is the best option. WTE emissions are lower if LCA system boundaries are expanded to include offsets for recovering scrap metals from WTE bottom ash.

Fruergaard & Astrup (2011) compare waste-to-energy technologies in Denmark.
Goal and scope: The goal was to compare two different waste-to-energy technologies (co-combustion in coal-fired power plants and anaerobic digestion) with mass burn incineration with and without energy recovery. The scope of the study included two different waste fractions: i) a high calorific fraction (SRF) suitable for co-combustion and ii) organic waste suitable for biological treatment. In total 7 different combinations of WTE technologies and waste fractions were examined.
Functional unit: utilization of 1 tonne of SRF/organic waste for energy purposes, including collection and pre-treatment.
LCI: data were collected from refereed literature and operation of incinerators in Denmark
Software used: EASEWASTE
Assumptions: production of capita; goods was not included as their impacts were assumed to be of minor importance per tone of waste throughout the life cycle of the plants
LCIA: Based on the EDIP 1997 method. The impact categories are: global warming, acidification, nutrient enrichment, photochemical ozone formation, human toxicity via soil, water and air, ecotoxicity in water and in soil.
Main conclusions: Overall, waste incineration with efficient energy recovery proved to be a very environmentally competitive solution based on Danish conditions. Co-combustion of SRF at modern power plants appeared fully comparable provided that sufficiently well flue gas cleaning systems are installed. Anaerobic digestion of organic waste materials appeared less preferable overall.

7. Conclusions

Based on the 21 references reviewed in the chapter, the following conclusions can be drawn:
LCA has been applied to various MSW management stages covering the whole MSW life cycle: 3 publications refer to collection (Rives et al., 2010; Iriarte et al., 2009; Beigl & Salhofer, 2004); 10 publications refer to integrated MSW management (Abduli et al., 2010; Miliūtė & Staniškis, 2010; Banar et al., 2009; Cherubini et al., 2009; De Feo & Malvano, 2009; Khoo, 2009; Liamsanguan & Gweewala, 2008; Buttol et al., 2007; Hong et al., 2006, Özeler et al., 2006); 6 publications refer to waste-to-energy schemes (Fruergaard & Astrup, 2011; Chen & Christensen, 2010; Moris, 2010; Wittmaier et al., 2009; Chaya & Gweewala, 2007; Wanichpongpan & Gweewala, 2007); Finally, there are 2 publications that deal with specific MSW streams: 1 for WEEE (Hischier et al., 2005) and 1 for the biodegradable fraction of MSW (Mendez et al., 2003).
Regarding the collection and storage of MSW, LCA revealed the following conclusions: smaller volume containers have the greatest environmental impact (Rives et al., 2010); HDPE containers have greater impact compared to steel (Rives et al., 2010); the multi container collection system has the least environmental impact while the door-to-door system has the greatest (Iriarte et al., 2009); kerbside collection is environmentally better than collection in the bring system (Beigl & Salhofer, 2004).
Coming now to the integrated MSW management, the following conclusions were identified: landfills are the worst management options (Miliūtė & Staniškis, 2010; Cherubini et al., 2009; Wanichpongpan & Gweewala, 2007; Hong et al., 2006; Mendes et al., 2003);

significant environmental savings are achieved from energy recovery (Fruergaard & Astrup, 2011; Cherubini et al., 2009; Khoo, 2009; Wittmaier et al., 2009; Liamsanguan & Gweewala, 2008; Buttol et al., 2007; Wanichpongpan & Gweewala, 2007); the same is true for material recovery, especially metals (Morris, 2010; Banar et al., 2009; Buttol et al., 2007; Özeler et al., 2006; Hischier et al., 2005); the selection of the best scenario depends on the impact category examined (De Feo & Malvano, 2009).

Finally, the waste-to-energy case studies, in addition to the aforementioned conclusions, reveal the following: energetic utilisation of waste with increased calorific value should be pursued (Wittmaier et al., 2009); the fluidized bed incineration without coal consumption saves more potential impacts than grate furnace incineration technology (Chen & Christensen, 2010); electricity from waste-to-energy incineration is not better than electricity from natural gas (Morris, 2010); waste incineration is preferable to anaerobic digestion for Fruergaard & Astrup (2011); however, the opposite is reported by Chaya & Geweewala (2007).

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Part 5

Leachate and Gas Management

25

Odour Impact Monitoring for Landfills

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1. Introduction

In the perspective of the improvement of life quality and citizens wellness, odour pollution is becoming a more and more relevant topic. In fact, among the variables that could influence the citizens' sense of a healthy environment, odour emissions play an important role, as they deeply affect the human life quality and psycho-physical wellness.

An odour is a mixture of light and small molecules, that are able to stimulate an anatomical response in the human olfactory system (Craven et al., 1996). The nose represents the interface between the ambient air and the central nervous system; in fact chemicals interact with the olfactory epithelium which contains different olfactory receptors and the signals are transmitted to the brain, where the final perceived odour results from a series of neural computations. The olfactory signals are processed also thanks to the memory effect of previous experienced smells, thus accounting for the high subjectivity of the odour perception (Freeman, 1991; Pearce, 1997).

In this way the sense of smell permits to detect the presence of some chemicals in the ambient air and for this reason odour perception is sometimes associated with a risk sensation (Dalton, 2003; Rosenkranz & Cunningham, 2003) or however it represents an indicator of a not salubrious situation for people suffering for olfactory pollution. Although odours do not involve toxic effects for human health, they could cause both physiological symptoms (respiratory problems, nausea, headache) and psychological stress (Schiffman, 1998). For this reason in the last decade the scientific community has been developing an increasing attention for odour pollution, generally caused by different types of industrial activities such as tanneries, refineries, slaughterhouses, distilleries, and above all civil and industrial wastewater treatment plants, landfills and composting plants. Moreover, the proximity of these industrial plants to residential areas really affects the acceptability of such activities causing population complaints (Nicell, 2009; Stuetz & Frechen, 2001; Yuwono & Lammers, 2004).

This paper focuses on the necessity of a proper management for odour emissions during the processes and the critical phases of landfills, and on the development of a proposal for a guideline to evaluate odour emissions and odour impact. So, the methodological approach of the guideline is described and compared with those commonly adopted in odour regulations.

2. Landfill odour emissions

Landfills are the most common way of disposing of municipal solid wastes (MSW).
Among the several existing types of industrial plants that generally cause odour nuisance, they represent one of the major sources of odour emissions and complaints.
Emissions from municipal landfill sites approximately consist of 65% v/v methane and 35% v/v carbon dioxide (Allen et al., 1997), while trace volatile organic compounds (VOC) represent less than 1% v/v of landfill gas. Odour emission is attributed to the presence of low concentrations of VOC, in particular esters, organosulphurs, alkylbenzenes, limonene, other hydrocarbons and hydrogen sulfide (Young & Parker, 1983).
Odour emissions originate principally from the atmospheric release of compounds deriving from biological and chemical processes of waste decomposition (ElFadel et al., 1997). In particular, the anaerobic degradation of the biodegradable fraction of the MSW causes several environmental problems such as methane and leachate production and VOC and odours emission (Scaglia et al., 2011).
The odorous characteristics of landfill gas may vary widely from relatively sweet to bitter and acrid, depending on the concentration of the odorous substances within the gas. These concentrations could be affected by several factors, such as the waste composition, in particular its organic fraction (OFMSW), the decomposition stage, the rate of gas generation and the nature of microbial populations within the waste. Moreover the weather conditions (wind speed and direction, temperature, pressure, humidity) significantly affect the extension of the area in which odours spread away from the landfill boundaries.
Generally the presence of OFMSW in landfills can be reduced by three different approaches (Scaglia et al., 2011):

- separation of OFMSW to produce compost;
- waste burning to produce energy;
- mechanical–biological treatment (MBT) (composting-like process) to produce a stabilized material.

The MBT is often carried out directly in landfill plants; it consists in a solid-state aerobic process (composting-like process) during which forced aeration in the biomass allows the microbial oxidation of the organic fraction of MSW, reducing its potential impact (Scaglia & Adani, 2008; Scaglia et al., 2010). In this process it is necessary to maintain the optimal aeration conditions in the biomass in order to avoid the production of intermediates of the anaerobic metabolism (e.g., sulphide and nitride compounds). In fact, odour emission mainly occurs during the first phase of the aerobic process when oxygen limitation for the aerobic biological process becomes more evident. Oxygen limitation could be due to both the high rate of O_2 consumption, because of the great amount of degradable organic matter present in the biomass, and to insufficient air diffusion.
However the main sources of odour emissions are represented by fresh waste dumps stored everyday. In order to reduce these emissions, it is opportune using cover materials after daily waste storage in landfills. Conventionally, materials deriving from the construction and demolition industry have been considered suitable to the purpose (Hurst et al., 2005), but other materials have been regarded as an alternative, such as paper mill sludge, fly ash, mulched wood material and foams (Bracci et al., 1995; Bradley et al., 2001; Carson, 1992; Hancock et al., 1999; Shimaoka et al., 1997). In the perspective of a sustainable waste management, the use of the stabilized materials derived from MBT process is deemed suitable for reducing odour emissions.

3. Odour emission monitoring and control

Odour emission monitoring and its regulation are characterized by a great complexity due principally to the strict association of odour pollution to human perception. For this reason, odour emission monitoring and its control can not be rigorously equalled to air quality monitoring.

Commonly, for air quality monitoring the conventionally used approaches are focused on:

- *impact evaluation and estimation of the pollutant relapse on the territory.* This aspect is generally attained by means of decision making support tools and, in particular, of dispersion models that estimate the downwind concentration according to emission rates, meteorological parameters, that affect the transport and the diffusion of the pollutants, and topography of the site. About odour emissions, dispersion models are considered a useful tool for predicting odour impact. However, there are some typical aspects that have to be taken into account when the modelling is performed for odour. First of all, odour is a mixture, composed by a lot of chemical substances, with different physical and chemical properties, that can react each other and change their composition. In a dispersion model, odour is considered as a pure substance rather than a combination of different chemicals. So, it is modelled as a single indicator compound, usually with a low odour threshold and a high emission rate (Drew et al., 2007).

  Moreover, in many cases the dispersion models are not suitable to describe the human odour sensation that is activated by the odour stimulus in few seconds (Schauberger et al., 2002). Odours therefore produce a response in the receptor quicker than other atmospheric pollutants (Irish Environmental Protection Agency, 2001). Furthermore odour emissions are discontinuous, alternating periods of high emission rate with periods of low emissions (Drew et al., 2007); greater annoyance is mainly caused by short periods of odour than by longer lasting odour emissions, as the olfactory sense is able to adapt to persistent odours, thereby reducing annoyance (Guideline on odour in ambient air [GOAA], 1999). For this reason, the fluctuations from the mean concentration, rather than the mean value, frequently affect the odour perception (Best et al., 2001). So, the average time used by dispersion models for the estimation of odour concentration represents another critical point.

  The dispersion models are normally based on long averaging time periods, usually 1 hour, whereas odours can generate community complaints from a series of short detectable exposures (Mahin, 2001; Mussio et al., 2001). The concentration values, predicted in this way, represent the concentrations of a mixed sample of ambient air that have been sampled over a 1-h period. Since meteorological conditions are highly variable over very short periods of time, the use of a 1 hour average masks the short-term peak odour concentrations that are experienced by the population (Nicell, 2009). However, 1 hour averaging time is also used because the most frequently available atmospheric input data are recorded as hourly averaged variables. An approach for overcoming this drawback involves the use of short averaging times for considering concentration peaks and thereby obtaining a more accurate prediction of odour dispersion. New generation air dispersion models can run at averaging times of less than 1 h, as half-hourly mean (Schauberger et al., 2002) or 10 – minute averages (Nicell, 2009), even if they are typically not used by regulators. Furthermore only few dispersion models are able to estimate short-term concentrations, while most models use highly simplified and uncertain methods to convert the commonly estimated one-hour average concentrations to short-term averages (Nicell, 2009; Schauberger et al., 2002).

- *monitoring through standard methodologies*. Air quality monitoring is commonly performed using conventional analytical methodologies that produce a list of substances involved and their concentration. Even for odour emissions, an instrumental approach, usually the conventional Gas Chromatography coupled with Mass Spectrometry (GC/MS) (Davoli et al., 2003; Dincer et al., 2006), is widely used in order to evaluate the odorous air chemical composition. Nevertheless the perceived odour results from many volatile chemicals, often at concentration lower than the instrumental detection limit, that synergically interact or add according to non predictable laws (Craven et al., 1996; Vincent & Hobson, 1998; Yuwono & Lammers, 2004). Furthermore the GC/MS is expensive and does not give information about human perception, thus not allowing a linear correlation between a quantified substance and an olfactory stimulus (Di Francesco et al., 2001).

  In fact, a reliable odour monitoring technique must be representative of human perception, trying to overcome the subjectivity due to the human olfaction variability and providing accurate and reproducible results. The more sensitive and broader range odours detector is undoubtedly the mammalian olfactory system; so, there is a growing attention for odour measurement procedures relying on the human nose as detector, in compliance with a scientific method (Craven et al., 1996; Pearce, 1997; Walker, 2001). So, dynamic olfactometry represents the standardized method for the determination of odour concentration; it is based on the use of a dilution instrument, called olfactometer, which presents the odour sample, diluted with odour-free air according to precise ratios, to a panel of selected human assessors. In the last years, the conventional instruments for chemical analysis (GC/MS) have been coupled with sensory detection developing a gas chromatography-olfactometry (GC-MS/O) technique in order to study complex mixtures of odorous compounds. GC-MS/O allows a deeper comprehension of the odorant composition in terms of compounds identification and quantification, offering the advantage of a partial correlation between the odorant chemical nature and the perceived smell (Friedrich & Acree, 1998; Lo et al., 2008).

  Both analytical and sensorial methods provide punctual odour concentration data and do not allow to perform continuous and field measurement, useful for monitoring odour emissions that can vary over the time in consideration of the industrial processes. To the purpose, artificial olfactory instruments (E – Noses) miming the biological system (Craven et al., 1996; Pearce, 1997; Peris & Escuder-Gilabert, 2009; Snopok & Kruglenko, 2002) have been developing. E-Noses are based on "an array of electronic-chemical sensors with partial specificity to a wide range of odorants and an appropriate pattern recognition system" (Gardner & Bartlett, 1994). The chemo-electronic signals are processed by pattern recognition techniques (i.e., artificial neural networks, multivariate statistical analysis) for their classification in order to identify odour and quantify the concentration. These systems present lower costs of analysis, rapidity of the results and allow to carry out continuous field monitoring nearby sources and receptors. After a training phase, electronic noses are able to preview the class of an unknown sample and consequently to associate environmental odours to the specific source.

In the following paragraphs the principal methodologies for odour monitoring (dispersion models, chemical characterization, dynamic olfactometry and chemical sensors) will be described, presenting their applications for landfill monitoring.

3.1 Dispersion models

Atmospheric dispersion models are computer programs that use mathematical algorithms to simulate how pollutants disperse in the atmosphere and, in some cases, how they react. Since it is impossible to use direct measurements to evaluate odour dispersion over a long range and/or make predictions, dispersion models are widely applied to odour investigation. The use of dispersion models is indispensable in the studies for authorization processes, evaluation of odour impact at the receptors and process control.

Dispersion models calculate odour concentration at ground level using emission data, meteorological data and orographic data.

Emission data can be determined analyzing samples, collected at each source of the plant, by dynamic olfactometry and then calculating the odour emission rates (Hayes et al., 2006; Sironi et al., 2010). The indispensable input meteorological data include wind speed, wind direction, air temperature and solar radiation in the study area over a long enough period (Hayes et al., 2006; Sironi et al., 2010). Orographic data are useful to take into account the effects of the topography on odour dispersion (Chemel et al., 2005).

Simulated concentrations at receptors can be processed to calculate parameters to be compared with reference limits, such as annual or daily average values expressed as concentration percentiles. The averaged odour concentration, calculated at each receptor, has to be compared with exposition criteria employing percentiles, that represent a distribution of concentration values. The choice of a percentile indicates a level of exposition to odour nuisance, since it represents a value below which a fixed percentage of observations falls. For example, the 98th percentile of one year hourly simulations is equal to 175 hours; this means that the 98th percentile of a series of values is the datum not exceeded by the 98% of the distribution values (Capelli et al., 2010; Romain et al., 2008).

Three main categories of atmospheric dispersion models are currently used: Gaussian, Lagrangian, and Eulerian (Dupont et al., 2006):

- Gaussian models are relatively simple statistical models describing the scalar plume downwind from a source point as a Gaussian-type curve. This kind of models are suitable for flat areas but not for areas with a complex orography (McCartney & Fitt, 1985). These are parametric models, because they calculate odour concentrations on the basis of a set of input parameters. Even if they introduce extreme simplifications of the phenomena, they are quite simple to apply, and so, widely used (Chen et al., 1998; Hayes et al., 2006; Holmes & Morawska, 2006; Wang et al., 2006).
- Lagrangian models deduce average concentration and deposition rates from the trajectories of numerous individual particles. The odour concentrations are calculated considering the random paths of single particles and require many simulations of particles paths to achieve good results. According to the Lagrangian approach, the virtual particles follow a prescribed wind field modified by turbulence, and the model computes their spatial trajectories. As they cannot calculate the flow characteristics themselves, these models require velocity and turbulence fields to be prescribed a priori, which is not possible in most heterogeneous, real-world situations (Holmes & Morawska, 2006; Kaufmann et al., 2003; Stohl et al., 1998; Stohl & Thomson, 1999).
- In Eulerian approaches the mean particle concentration is directly calculated by solving the advection-diffusion equation in a tridimensional reference grid. Thus, the Eulerian approach is simpler than the Lagrangian one. These models are generally applied on mesoscale or urban scale, especially in the presence of complex chemical reactions. CFD

(Computational Fluid Dynamics) models have been developed in Eulerian framework for predicting flow and transport processes in urban or industrial environments taking into account the effects caused from buildings presence (Holmes & Morawska, 2006).

Furthermore puff models have been developed in which the pollutant is assumed to be emitted as a large number of puffs in rapid succession. They are non-stationary in time. This kind of models can be applied on wide domains or areas with complex orography (Holmes & Morawska, 2006; Wang et al., 2006).

Dispersion models are generally used in conjunction with other odour monitoring techniques to evaluate the landfill odour impact at the surrounding areas (Li, 2003; Romain et al., 2008) and to analyze the variation in odour exposure within communities surrounding landfill sites (Sarkar et al., 2003).

3.2 Chemical characterization

Chemical analysis of odour samples is able to provide the chemical composition of the single compounds in a mixture and their concentrations. Generally, characteristic compounds generating odours in a landfill are ammonia, hydrogen sulfide and VOC (volatile organic compounds) like amines, mercaptans, sulfur compounds, saturated and unsaturated fat acids, aldehydes, ketones, hydrocarbons, limonene, chlorinated compounds, alcohols, etc. (Bruno et al., 2007; Capelli et al., 2008; Dincer et al., 2006; Leach et al., 1999; Ribes et al., 2007).

VOC samples are collected using canisters (Camel & Caude, 1995; Kumar & Viden, 2007; Ras et al., 2009), polymer bags (Dincer et al., 2006; Ras et al., 2009) or adsorbent materials (Ras et al., 2009). Adsorbent materials, packed in appropriate tubes, represent a handier sampling method than canisters and bags because they allow to sample a great volume of air reducing the analytes in a small cartridge. The critical point is the choice of adsorbents (usually porous polymers or activated carbon, graphitized carbon black and carbon molecular sieves) (Camel & Caude, 1995; Gawrys et al, 2001; Harper, 2000; Matisová & Škrabáková, 1995) that depends on the chemical feature of compounds to be sampled (Kumar & Viden, 2007). A combination of different adsorbents is preferred to sample a wide range of compounds without breakthrough problems (Harper, 2000; Wu et al., 2003).

Sampling on adsorbent materials can be applied in "active"mode (defined volume of sample air pumped at a controlled flow-rate) or "passive" mode (without the use of a pump but through direct exposure to the atmosphere) (Bruno et al., 2007; Gorecki & Namiesnik, 2002; Seethapathy et al., 2008). For both procedures the analytes can be recovered through thermal desorption or liquid extraction (Bruno et al., 2007; Demeestere et al., 2007, 2008; Ras et al., 2009; Ras-Mallorquì et al., 2007). After sampling, preconcentration techniques are required: gas-solid enrichment using adsorbent materials, solid phase micro extraction (SPME), cryogenic preconcentration and purge and trap (Davoli et al., 2003; Demeestere et al., 2007; Ras et al., 2009). Since odours are complex mixtures of volatile organic compounds, in the gas-chromatographic analysis of odour samples critical steps are the choices of the appropriate column and detector to achieve a simultaneous determination of as much compounds as possible (Demeestere et al., 2007; Ribes et al., 2007; Zou et al., 2003).

Nevertheless, it is very difficult to establish a correlation between analytical measurements and odour intensities perceived, especially because of the different interactions between odourants in a mixture.

Example of applications of chemical characterization for landfill monitoring. Not many studies have been carried out on chemical characterization of odours in ambient air at a landfill site. Davoli et al. have analyzed air samples from different landfills using SPME and GC-MS to better establish specific markers of olfactory pollution (Davoli et al., 2003). Dincer et al. have investigated the composition of odorous gases emitted from a municipal landfill to find a relationship between odour concentration and chemical concentration of VOC by GC-MS (Dincer et al., 2006; Dincer & Muezzinoglu, 2006). Ambient air monitoring has been conducted at landfills using thermal desorption and GC-MS determination of VOC to identify the compounds responsible of potential odour nuisance (Capelli et al., 2008; Leach et al., 1999; Ribes et al., 2007; Zou et al., 2003).

3.3 Dynamic olfactometry

Nowadays the dynamic olfactometry is the standardized method used for determining odours concentration and evaluating odour complaints (CEN, 2003; Schulz & van Harreveld, 1996). Dynamic olfactometry is an instrumental sensory technique that employs the human nose (a panel of human assessors) in conjunction with an instrument, called olfactometer, which dilutes the odour sample with odour-free air, according to precise ratios, in order to determine odour concentrations.
Odour sampling is carried out using odourless containers and sampling lines. In particular, for olfactometric analysis, polymer bags of Tedlar® (polyvinyl fluoride) or Nalophan® (polyethylene terephthalate) are used for the collection of odorous compounds.
For samples of ambient air or punctual emissions, odour bags are filled using a depression pump that works on the basis of the `lung' technique: the bag is placed inside a rigid container evacuated using a vacuum pump (AS/NZS, 2001; ASTM, 2004; CEN, 2003).
In the case of areal sources, instead, it is necessary to use auxiliary devices to collect odour samples, because it is difficult to cover the whole emission area during sampling and so representative sampling sites have to be established (Bockreis & Steinberg, 2005). The investigations are conducted using a hood or a wind tunnel, depending on the measurement conditions. For olfactometric analysis the examiners are selected in compliance with a standardized procedure, performed using reference gases; only assessors who respect predetermined repeatability and accuracy criteria are selected as panelists.
Commonly, there are two standardized methods for the presentation of odour samples to panel: forced choice and yes/no method (AS/NZS, 2001; ASTM, 2004; CEN, 2003). In the forced choice method, two or more sniffing ports are used; the odour sample is presented at one port, and neutral air at the other port(s). In this case, the examiners have to compare the different presentations and to choice the port from which odour exits. In the yes/no method each examiner sniffs from a single port and communicates if an odour is detected or not. Odorous sample diluted with neutral air or only neutral air can exit from the sniffing port.
The process continues until each panelist positively detects an odour in the diluted mixture; at this stage the panelist has reached the detection threshold for that odour (AS/NZS, 2001; ASTM, 2004; CEN, 2003).
Since odour perception is a logarithmic phenomenon (Stevens, 1960), in this type of measurements it is necessary taking into account that odour concentration is associated to odour intensity though a defined logarithmic relation.
The concentrations may be expressed as threshold odour numbers (TON) or dilution to threshold (D/T) ratios. Although the concentrations are dimensionless, it is common to consider them as physical concentrations, and to express them as odour units per cubic meter (ou/m^3) (Frechen, 1994; Koe, 1989).

Although dynamic olfactometry represents the standardized objective method for the determination of odour concentration, it is affected by some limitations. First of all it provides punctual odour concentration data, but it is not sufficient to evaluate completely a case of olfactory nuisance, because it does not allow to perform continuous and field measurements, useful for monitoring the industrial processes causing odour emissions. Moreover, dynamic olfactometry considers the whole odour mixture and do not discriminate the single chemical compounds and their contribution to the odour concentrations. Odour samples are difficult to store, because of their instability, and so require rapid time of analysis. Finally, as it is well-known, olfactometry is too time-consuming and quite expensive and moreover frequency and duration of analysis are limited.

Example of applications of dynamic olfactometry for landfill monitoring. Olfactometric measurements have been employed by Sironi et al. (Sironi et al., 2005) for sampling the principal odour sources of seven Italian landfills in order to estimate an Odour Emission Factor (OEF). Sarkar et al. have used olfactometric analysis on samples from various sensitive areas of a municipal solid waste (MSW) landfill site to find a relationship between odour concentration and odour intensity (Sarkar & Hobbs, 2002). Many authors have carried out odour measurement at landfills using more than one technique to characterize such complex plants; so dynamic olfactometry has been coupled with GC-MS analysis (Capelli et al., 2008; Pagè et al., 2008), electronic nose (Capelli et al., 2008; Li, 2003; Romain et al., 2008; Snidar et al., 2008), dispersion modelling (Li, 2003; Snidar et al., 2008), odour patrol monitoring (Li, 2003) and field determination (Romain et al., 2008).

3.4 Chemical sensors

The need of a more analytical approach to the quantitative measurement of odours has led to the use of chemical sensors. The response of the chemical sensors with partial selectivity is measured upon exposure of the sampled odour or multicomponent gas-mixture and the pattern based on the overall response of a sensors array defines a chemical fingerprint related to a given sampled odour. The recorded data of the sensors array response towards various odours can be usually processed by pattern recognition techniques (i.e., artificial neural networks, multivariate statistical analysis) for their classification, in order to identify an odour and quantify the concentration.

Chemical sensors are integrated with a sampling system and a signal processing unit to have an electronic nose (E-Nose), that is a device developed to reproduce the human olfactory system. An E-Nose requires a training for any specific application, but, on the other hand, it is a rapid and economic alternative to other techniques of odour measurement. The type of chemical sensors which can be used in an E-Nose need to be responsive to molecules in the gas phase. At present, there are two main types of gas sensor: metal oxide (MOX) and conducting polymer (CP) resistive sensors.

Gas sensors, based on the chemical sensitivity of semiconducting metal oxides, are readily available commercially and have been more widely used to make arrays for odour measurement than any other single class of gas sensors. A deep overview on sensor materials for odour detection can be found in literature (Gopel et al., 1992; Sbreveglieri, 1992).

All chemical sensors comprise an appropriate and chemically-sensitive material interfaced to a transducer, hence, the solid-state sensors are essentially constituted by a chemically sensitive interface (sensitive material) and a transducer. The classification of chemical sensors can be realized on the basis of the transducer used: conductometric (resistive), optical, electrochemical, mechanical/acoustic or ultrasonic, thermal, MOSFET (metal–oxide–semiconductor field-effect transistor) (D'Amico & Di Natale, 2001; D'Amico et al., 1995). A transduction process is realized by converting the input-event or measurand into an output electrical signal (analogue voltage or current, digital voltage) correlated to the measurand that generates it. The output electrical signal is properly conditioned, processed and stored for analysis.

Example of applications of chemical sensors for landfill monitoring.

Many multiparameter portable sensor-systems have been studied and exploited in field measurements for air quality control of toxic pollutants (NOx, CO, SO_2, H_2S) (Al-Ali et al., 2010). Moreover, sensors arrays have been used for odour monitoring of landfill municipal sites and for odour quantification (Micone & Guy, 2007; Nicolas et al., 2000; Persaud et al., 2005, 2008; Penza et al., 2010). Comini et al. have tested solid-state chemoresistive gas sensors based on mixed-oxides thin films to detect odourous compounds in gases produced by a landfill (Comini et al., 2004). Perera et al. implemented an electronic nose with a small computer to easily provide remote control of bad odours in landfill sites (Perera et al., 2001). Persaud et al. (Persaud et al., 2008) used a single-point E-Nose instrument for continuous monitoring of odours in the biogas produced by wastes fermentation along the perimeter of a municipal landfill site. Since landfills represent one of the major causes of odour complaints (Sironi et al., 2005) and a kind of plant difficult to monitor, because of the great variety of substances that may cause odour nuisance, the use of more than one technique for odour determination is required. For a complete characterization of odours, Capelli et al. have used olfactometry, chemical analyses with GC-MS and electronic noses, finding that even if the results of the three different odour characterization techniques do not necessarily correlate, each one contributes to solve the complexity of odour measurement in the environment (Capelli et al, 2008). Other comprehensive investigations on landfill areas used olfactometry in combination with: dispersion modelling, odour patrol monitoring and E-Nose (Li, 2003); dynamic olfactometry, field determination of odour perception points and electronic noses (Romain et al., 2008). Another approach was carried out using results of olfactometric analysis as input for a dispersion model and E-Noses for continuous monitoring to determine the landfill odour impact on a specific receptor (Snidar et al., 2008).

4. A methodological approach for the definition of an odour guideline

4.1 Odour regulations: the principal approaches

The odour emission regulation is generally tackled valuating two aspects:

- *emissions*, expressed as the odour concentration released by a particular source. In this case, two approaches have been adopted by the legislations of the different countries, establishing precise limits for the whole odour mixture and/or for single chemical compounds. In the first case, the odour concentration is expressed in odour units (ou/m^3) and detected through dynamic olfactometry. In the second one only the concentrations of specific compounds are set, expressed in typical mass/volume units; the limits have been established based on odour thresholds rather than toxicological impacts (RWDI Air Inc., 2005). Such odour limits are related with compounds that have a typical odorous impact (e.g., ammonia, hydrogen sulphide, methyl mercaptans) (Nicell, 2009).

Because of the wide range of odour industrial processes and sources (punctual or active/passive areal sources), the prescriptive limits for the odour mixtures usually refer to specific sources (above all punctual or active areal sources) and to precise plants (above all composting plants). In particular, for passive areal sources, such as dumps in landfills, it is extremely critical fixing limits, due to the variability of the amount of stored materials and of the area extension.

- *odour impact criteria*, defined as odour concentration limits considered acceptable for avoiding odour annoyance at receptors. They are typically expressed in terms of a concentration (i.e., in ou/m^3) considering an averaging time and a frequency of exposure (e.g., 98^{th} percentile of hourly average concentrations in one year). Odour concentrations at receptors are estimated using appropriate dispersion models, that determine whether the emissions are in compliance with odour impact criteria. These limits have a predictive nature and establish very low odour concentrations that are not detectable by available measurement methodologies.

Both the aspects present some limitations, such as the difficulty of assigning an emission limit because of the wide range of odour sources and/or the complexity of choosing the opportune parameters for models. So, it seems necessary to consider an integrated approach in order to overcome these drawbacks.

4.2 An approach for an odour guideline

The actual approaches for odour regulation do not adequately satisfy the requests of monitoring and control expressed by the population directly exposed.

In this section, a proposal of an odour guideline is presented with the purpose of defining acceptability and monitoring criteria for odour emissions produced by landfills.

According to the principle of pollutant prevention and reduction, commonly adopted by environmental legislations, the present methodology suggests a coupling between a predictive approach, based on dispersion models, and a systematic approach to carry out the monitoring and the control through reliable methodologies.

The acceptability criteria, resulting from the odour guideline proposal, will be verified and applied taking into account the background values detected on the territory. Such background levels could be assessed by means of ad hoc measurements (in appropriate meteorological conditions or when sources are not active) carried out by dynamic olfactometry. In particularly complex cases, such as co-presence of other significant odour sources, this evaluation could be executed through chemical characterization of ambient air samples. In addition, depending on the plant odour impact, an appropriate continuous monitoring system should be planned in order to perform a processes control.

The focus of the proposal consists in the implementation of two approaches for the authorization of odour emissions:

- assessment of acceptability criteria using predictive methods;
- the buffer zone approach.

4.2.1 Assessment of acceptability criteria (Y) using predictive methods

This approach employs the use of mathematical models to predict the downwind odour concentrations at receptors on the basis of odour emission rates, topography and meteorological data referred to a selected period of time. Such models aim to determine whether the estimated emissions at sensitive receptors are in compliance with ambient air quality criteria, considered acceptable for the exposure of the population. These criteria, named Y, can be defined on the basis of several parameters, such as:

- presence of sensitive receptors;
- distance between the plant and sensitive receptors;
- land use (residential, commercial, agricultural, industrial);
- existing or new plants;
- distribution of concentration values expressed as percentiles;
- averaged time considered for simulations.

So, Y is set as a specific percentile value related to an averaging time calculated through dispersion models.

For example, assuming that Y is equal to 2 ou/m^3 as 98th percentile of hourly average concentrations in one year at the first receptor, figure 1 and figure 2 illustrate modelling simulations executed for a landfill. In figure 1 the receptor is situated in an area where the predicted odour concentration is lower than 2 ou/m^3, so the Y limit is fulfilled; on the contrary, figure 2 shows the case in which Y is not attained at the receptor.

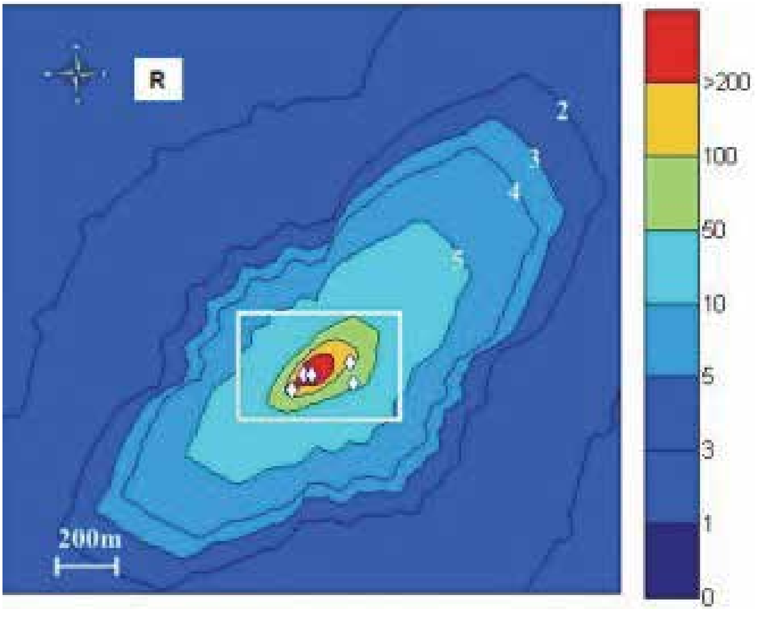

Fig. 1. Map showing $Y_{predicted}$ in compliance with Y_{limit}. The white rectangle delimits the landfill perimeter; the white points inside the plant represent the odour sources. R indicates the position of the receptor.

According to the ratio between the limit value and the predicted one ($Y_{predicted}$ / Y_{limit}), the implementation of continuous monitoring systems should be planned in order to perform a process control, at the boundaries of the plant and/or at the sensitive receptors, useful for identifying critical phases, under different meteorological conditions. Figure 3 shows the case in which, at receptor, the ratio between $Y_{predicted}$ and Y_{limit} is greater than an acceptable value (e.g. 0.80). The figure illustrates the implementation of three monitoring systems, positioned at the plant perimeter, according to the predominant wind directions, and at the receptor.

The output data of sensors or analyzers used for continuous monitoring must show a correlation with odour concentration, expressed in odour units.

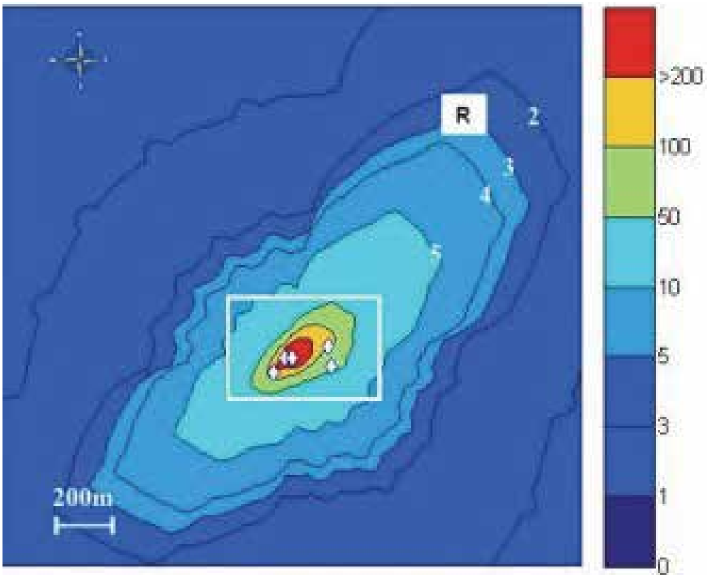

Fig. 2. Map showing $Y_{predicted}$ not in compliance with $Y_{limit.}$ The white rectangle delimits the landfill perimeter; the white points inside the plant represent the odour sources. R indicates the position of the receptor.

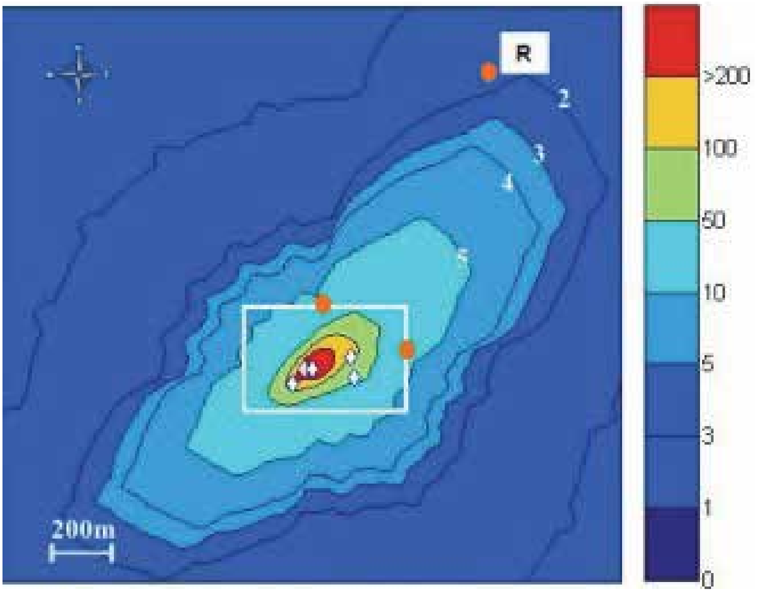

Fig. 3. Individuation of continuous monitoring points. The white rectangle delimits the landfill perimeter; the white points inside the plant represent the odour sources. R indicates the position of the receptor. The red circles represent the continuous monitoring systems.

4.2.2 The buffer zone approach (Z)

The buffer zone identifies an area around the plant boundaries, outside of which a prescriptive limit, named Z, expressed in odour units and detectable through dynamic olfactometry, must never be exceeded. Furthermore, in the area between the plant perimeter and the buffer zone boundaries a maximum concentration value, named X, must never be exceeded. The buffer zone can have a more or less regular shape, individuated according to the predominant wind directions, the presence of receptors and the geographic location. The buffer zone extension can be defined using dispersion models based on the meteorological scenarios that have determined the worst odour dispersion conditions in a defined period of time. These scenarios have to be described so that a possible exceeding, determined by a meteorological situation worse than those previously considered, could be permitted. If the Z limit is fulfilled inside the plant boundaries, the buffer zone overlaps with the plant perimeter.

Figure 4 and figure 5 explain how the buffer zone is defined according to the worst scenarios. For example, if Z is equal to 50 ou/m^3, the buffer zone must comprehend the area where 50 ou/m^3 are overcome in the worst meteorological conditions. In figure 4 (case 1) the buffer zone is outside the plant perimeter, while in figure 5 (case 2) overlaps with it. In this last case, the Z value must be applied and verified at the plant perimeter.

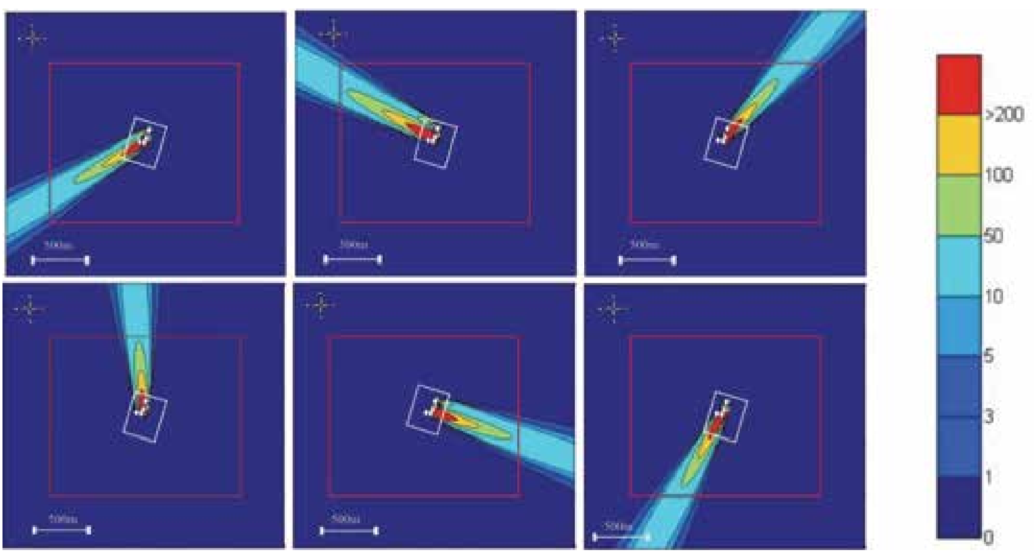

Fig. 4. Maps illustrating the individuation of the buffer zone considering the worst odour dispersion conditions for a landfill (case 1). The white rectangle delimits the plant perimeter while the red one individuates the buffer zone perimeter; the white points are the odour sources. In all maps, the buffer zone is defined on the bases of the isoline of 50 ou/m^3.

The definition of a buffer zone is a valid approach particularly for landfills that present areal emissions, usually located in ground-line; in fact, in this type of emissions the odour concentration decreases moving away from the sources. Z and X values must be verified by means of olfactometric measurements of ambient air samples. For this purpose, a monitoring plan should be proposed and verified. An example of monitoring plan is presented in Figure 6 where the six sampling points, located at the buffer zone perimeter, have been chosen upwind and downwind according to the predominant wind direction detected during the sampling. The other sampling points within the buffer zone have been located moving away from the plant boundaries along the predominant wind direction.

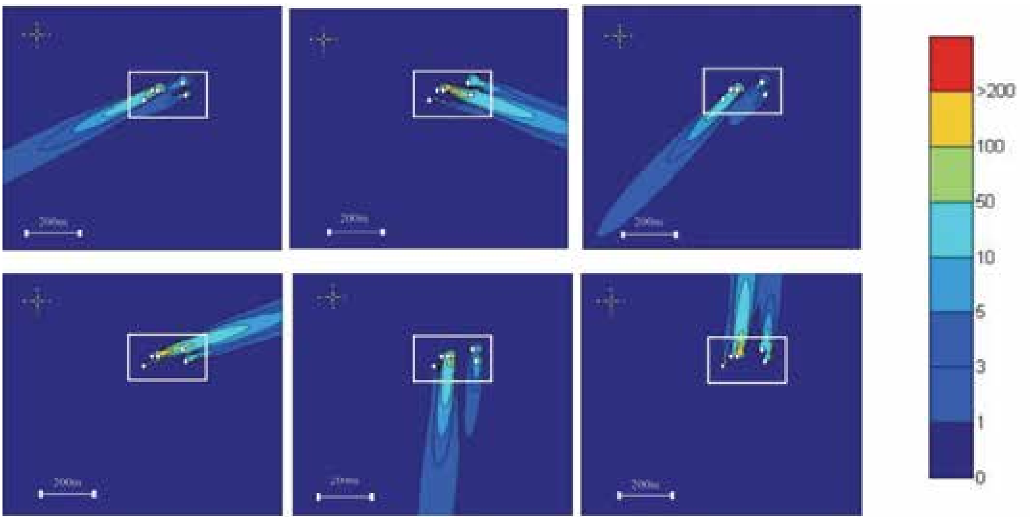

Fig. 5. Maps illustrating the individuation of the buffer zone considering the worst odour dispersion conditions for a landfill (case 2). The white rectangle delimits the plant perimeter; the white points are the odour sources. In all maps, since the isoline of 50 ou/m^3 falls within the plant perimeter, the buffer zone overlaps with the plant boundaries.

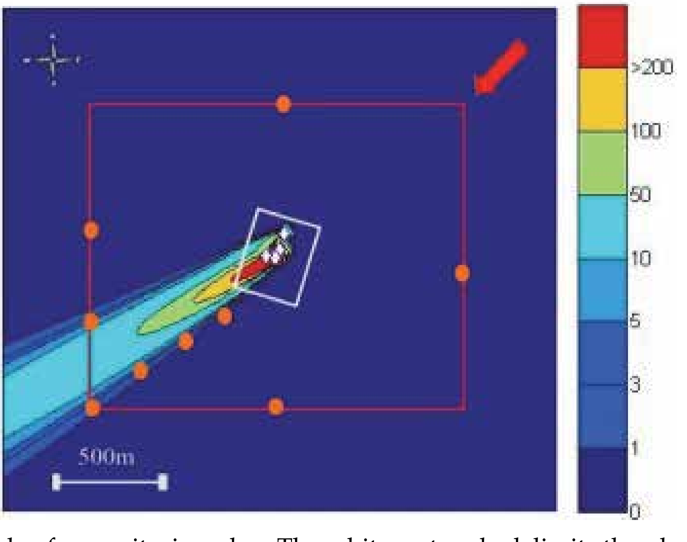

Fig. 6. An example of a monitoring plan. The white rectangle delimits the plant perimeter while the red one individuates the buffer zone perimeter. The red arrow shows the predominat wind direction; the orange circles indicate the location of the sampling points.

As described for Y limit, the implementation of continuous monitoring systems can be planned in relation to the extension of the buffer zone and the presence of receptors inside

or near it. According to the different conditions, these systems can be situated at the receptors and/or at the boundaries of the buffer zone and of the plant.
Figure 7 shows an example for the localization of the monitoring systems; in this case the systems have been positioned at the receptors and at the boundaries of the buffer zone considering the predominant wind directions.

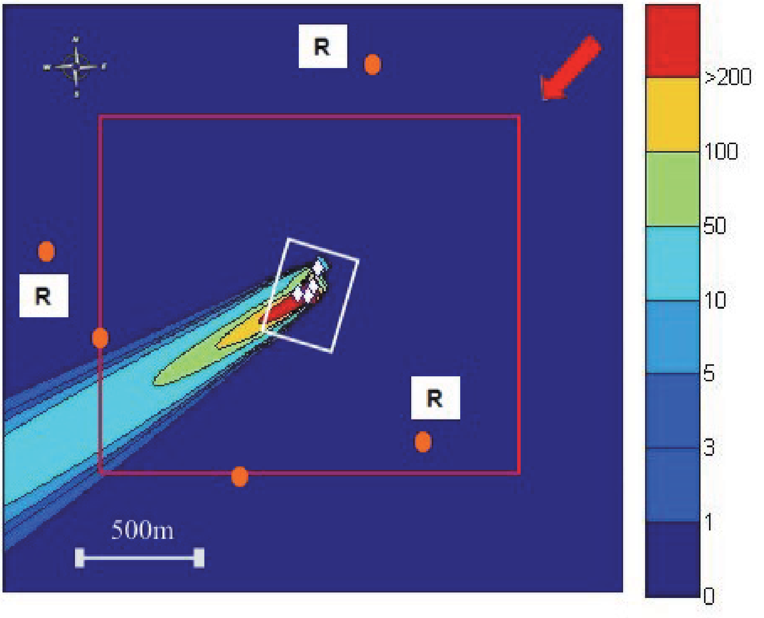

Fig. 7. An example of the localization of continuous monitoring points. The white rectangle delimits the plant perimeter while the red one individuates the buffer zone perimeter. The red arrow shows the predominat wind direction; the orange circles indicate the continuous monitoring systems. R indicates the position of the receptors.

5. Conclusions

The increasing attention of the population to olfactory nuisance and the proximity of industrial plants to residential areas have created the need of evaluating odour impact and regulating odour monitoring and control. Nowadays the adopted regulations do not adequately satisfy the requests of monitoring and control expressed by the population directly exposed.
In this paper, a proposal of a guideline for assessing landfill odour impact has been described; the guideline integrates a predictive approach based on dispersion models and a systematic approach to carry out the monitoring and the control. The novelty of the proposal is represented by the introduction of a buffer zone, individuated by means of dispersion models, in which prescriptive limits have to be fulfilled and verified by standard measurement methodologies. In addition, the odour guideline recommends to perform a process control for particularly impactful plants, realized through continuous monitoring systems. This methodological approach can be easily adopted even for the regulation of other industrial activities that cause odour emissions.

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26

Performance Indicators for Leachate Management: Municipal Solid Waste Landfills in Portugal

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1. Introduction

The significant concern of leachate management refers essentially with water, groundwater, and soils pollution, determining the need of adequate treatment for discharged in water, soil, or wastewater collection networks.

Leachate generation is an inevitable consequence of landfill waste disposal. An adequate landfill leachate collection and treatment allows proper environmental protection and prevention of surface water, groundwater, and soils contamination. It also minimises operational costs in the overall landfill management. The design and construction of leachate treatment plants strongly depends on the quality and quantity of the raw leachate, which in turn is influenced by numerous factors, including rainfall, waste composition, age of fill, landfill design and construction and operational procedures (Qasin e Chiang, 1994).

A detailed knowledge of local leachate management and treatment allows the identification of specific setbacks and constrains that need resolution, on an operational management view as well as with legal decisions to be made for leachate treatment performance.

In 2008, an assessment on the status of Portugal mainland's Municipal Solid Waste (MSW) landfill leachate management was developed (Martinho et al., 2008, 2009). The main objectives of this study intended to (Martinho et al., 2009):

- Evaluate current status on leachate generation and treatment in MSW landfills;
- Develop and apply performance indicators and other relevant context information that enables benchmarking analysis, regarding leachate treatment plants;
- Identify and determine constraints (i.e. environmental, operational, and economical) related to leachate generation and treatment on a national context and possible minimisation measures;
- Identify practices and tendencies in the field of leachate treatment technologies mainly in other EU Member States, and application on a national basis.

Performance indicators have been developed for water and wastewater services. (Alegre et al., 2004; Matos et al., 2004). The structure was designed to assess the performance of Management Entities (ME) that provide these services, in terms of their activities and intervention areas. Regarding waste management, Cunha and Simões (2010) describe the existing performance indicators used by the Portuguese Water and Waste Regulatory

(IRAR), for monitoring service quality of the wholesale segment of the waste service in Portugal. These performance indicators allow IRAR to develop a benchmarking analysis, on an annual basis, for the regulated waste ME.
In the matter of leachate management and consequent proper environmental protection, performance indicators and other relevant context information may be a valuable tool. In this case, one of the activities of the MSW ME is subject to benchmarking analysis: treatment and management of leachates generated in MSW landfills, in contrast to the overall performance of the operators.
In this chapter, a set of performance indicators (e.g. environmental, economical, operational, human resources, service quality, and opinion) for MSW landfills leachate treatment and management are defined and proposed. Considering the Portuguese case, a background on national MSW landfill and leachate management is given. The results obtained with the proposed performance indicators, as well as relevant context information, applied to Portuguese ME and leachate treatment facilities, will be presented in detail. Possible future directions in landfill operation and leachate treatment technologies to be applied will also be discussed.

2. Leachate management: municipal solid waste landfills in Portugal

Landfilling is the terminal operation of the waste management system, where non recyclable waste or waste that cannot be subject to valorization, is eliminated through deposition above or below the land surface. It is an essential component in any waste management system. The efforts for reduction, reuse, valorization, recycling and incineration can reduce waste quantities, however residual materials still remain and need adequate final destination.
In the European Union, the Landfill Directive (Directive 1999/31/CE of the Council, 26 of April) defines legal framework on landfills. It establishes measures, processes and guidelines that avoid or reduce, as possible, the negative effects on the environment, especially regarding surface water, groundwater, soil and air pollution, as well as the risks posed by these effects on human health, resulting from landfill disposal (EC, 1999).
In Portugal, the first Municipal Solid Waste Strategic Plan (PERSU I) was published in 1997. It established the main strategic guidelines that allowed the eradication of the 341 open dumps referenced in 1995. It also promoted the construction of landfill infrastructures providing adequate MSW final destination. The Landfill Directive was transposed to the Portuguese law in 2002 by the Decreto-Lei n. º 152/2002. Figure 1 shows the evolution of the number of landfills and open dumps existing in Portugal, between 1996 and 2007 (Portuguese Environment Ministry [MAOTDR], 2007).
In 1996, 13 landfill facilities already respected part of the guidelines of the Council Directive proposal on waste disposal in landfills (97/C156/08). With the publication of Decreto-Lei n. º 152/2002, in 2002 all open dumps were closed and 37 landfills were operating according to Landfill Directive's specifications.
In 2007, five million tonnes of MSW were generated in Portugal. In the start of the second Strategic MSW Management Plan (PERSU II) that defines the existing mechanical biological treatment plants amplification and the construction of new plants in the next years, the quantities landfilled still achieved 63% of the generated MSW (IRAR and Portuguese Environment Agency [APA], 2008).
In terms of MSW management models, by the time the first strategic plan was published, in 1997 (PERSU I), there were 40 MSW management systems, where 11 were managed by

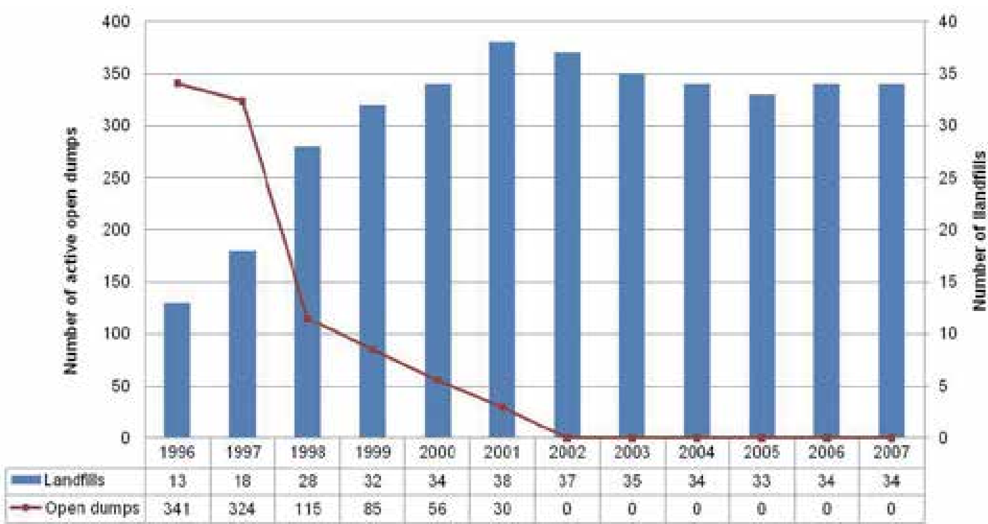

Fig. 1. Evolution of number of landfills in operation and active open dumps in Portugal between 1996 and 2007

concession entities (i.e. owned by public entities and the local municipalities) and 29 were managed by the local municipalities. Currently there are 29 MSW ME responsible for 14 closed sanitary landfills and 35 sanitary landfills in operation, all generating relevant quantities of leachate.

Considering the significant evolution of the national waste management system and the effective consolidation of the hierarchy principle for waste management options, the landfills legal framework was revised by the Decreto-lei n. º 183/2009. National legislation on landfill disposal obligates control and monitoring processes on leachate, groundwater, superficial water and leachate basins, both in the landfill operational and post-closure phases. Besides imposing on landfill operators the existence of leachate collection, adequate treatment, and final disposal, it also determines monitoring parameters and frequency for leachate, groundwater and surface water control in all landfill phases.

Discharge of leachate in streams, lakes, and soils is defined by Decreto-Lei n.º 236/98. It establishes norms for domestic and industrial wastewaters discharge to superficial and coastal waters, groundwater and soil, as well as to wastewater sewer systems.

In terms of legal requirements related to Integrated Pollution Prevention and Control (IPPC), all non-dangerous landfill operators, should obtain an Environmental License (EL) defined by Decreto-Lei n.º 194/2000. The EL defines that each operator must proceed with landfill leachate control, determining emission limit values for discharge on water and collection systems, with type and monitoring frequency. Limit values are defined for each infrastructure, although with respect to the definitions on Decreto-Lei n.º 236/98 and with the monitoring and control processes defined by landfill legislation.

3. Methodology

3.1 Performance indicators for leachate treatment and management

The performance indicators defined were based on the structure developed for water and wastewater services (Alegre et al., 2004; Matos et al., 2004). This structure was designed to assess the performance of ME that provide these services, as the overall performance of their activities and intervention areas. The intent of this work was more restrict. It aimed to assess one of the activities of the MSW ME: the MSW landfill leachate management, in contrast with the overall performance of the management entities. However not specifically defined as performance indicators, opinion indicators were added with intention to translate managers and technicians' perception about LTP operation and performance.

Relevant context information on ME and LTP were also included to have a main outline of their characteristics in the overall performance indicators. The adopted performance indicators groups for the purpose of this study are described as follows.

3.1.1 Environmental indicators (iAmb)

Indicators included in this group allow the assessment of the ME regarding the environmental impact of Leachate Treatment Plants (LTP). In terms of leachate treatment and discharge, indicators were defined by the percentage of disposal facilities where generated leachates are treated on-site in LTP or at Public Works Treatment Plants (PWTP) and percentage distribution of landfills by leachate discharge (i.e. water lines and municipal wastewater collection systems). The conformity with legal environmental monitoring and control and discharge norms were also considered, in terms of percentage of unconformities (i.e. raw leachate, groundwater and superficial water monitoring, leachate basins and other monitoring requirements), as well as the percentage of operational and environmental landfill licences. Indicators regarding treated leachate reutilisation, production and disposal of sludge and concentrates from membrane treatments were also considered relevant. In addition, indicators of upstream conditions that could influence LTP operation were considered: leachate production per landfill area, per landfilled waste volume and weight, annual leachate production per precipitation volume and biodegradable waste fraction in landfilled waste. For this group 22 indicators were defined.

3.1.2 Human resources indicators (iHR)

In terms of human resources, eight indicators were defined to assess human resources characteristics that are directly affected to LTP operation and maintenance, namely: number per volume of treated leachate and population equivalent, percentage of gender, qualifications (i.e. graduate education, secondary or other qualifications), operators percentage with specific education in leachate treatment and formation actions, in hours per year, by operator.

3.1.3 Operational indicators (iOP)

This group of indicators aims to assess the performance on operation and maintenance activities. It includes the percentage of leachate recirculation, storm discharge frequency, frequency of damages/problems per year (i.e. operational, logistics, personnel and other) parameter analyses performed (in percentage of the legally required) to leachate, groundwater and superficial water; number of maintenance inspections per year and water and energy consumptions per volume of treated leachate. A total of 12 indicators were proposed in this group.

3.1.4 Financial and economic indicators (iEF)

With the objective of assessing financial and economic considerations with leachate treatment and management, 15 indicators were defined. These include current expenses and capital unit costs, by leachate volume and population equivalent, unit investment, expansion or substitution unit investment, amortisations in the year of reference, costs fraction by personnel, energy, and other current expenses costs components.

3.1.5 Service quality indicators (iSQ)

In the particular case of leachate management and aiming to assess the performance of ME's leachate management quality of service, five indicators were included in this group, mainly referring to leachate treatment efficiencies and conformity with discharge limits.

3.1.6 Opinion indicators (iOpin)

With the intention to translate managers and technicians' perception about LTP operation and performance, six indicators were defined. A 5-point Likert scale was used to determine the opinion related to leachate treatment processes level of adequacy to raw leachate volume and quality, removal needs in terms of discharge option (water line or wastewater collection system), daily activities of LTP operation, number of operators adequacy and overall perception of LTP operation.

3.1.7 Context information

Aiming to account specific ME characteristics in the overall performance indicators, 28 context information variables were considered. This group included the total number of disposal facilities and distribution by operational landfill, closed landfills and old dumps existing within the ME intervention area, as well as the distribution in terms of surface area, waste volume and number of LTP. This information is relevant for a better framework on the ME.

On the other hand, information regarding disposal facilities, namely disposed waste volume and weight, percentage of biodegradable waste disposed and average annual precipitation were considered necessary variables for several performance indicators. Other information regarding landfill age, and conformity with the original design were also considered relevant for the overall analysis.

3.2 Questionnaire survey

For the purpose of application of performance indicators proposed to the Portuguese context, the analysis instrument used was a questionnaire survey to all MSW ME that managed MSW landfills LTP (i.e. 28 of 29). The objective of this survey was to collect relevant information related with the ME, quantity and origin of MSW landfilled, landfill characteristics, quantity, and quality of leachate and landfill gas generated, type of leachate treatment and final destiny.

The adopted logic for questionnaire development was broadness and flexibility, in order to adapt to MSW management models and to ensure a global and comparative result analysis. To ease questionnaire answer by the ME and improve response rate, given the extensive information to be gathered, the questionnaire was divided in two documents: Questionnaire1 and Questionnaire 2. The first was developed to collect more context information and general on ME, characteristics on MSW landfills where leachates are generated, including rainfall historical registry, type of leachate treatment and final destiny. The second questionnaire was prepared to gather more specific and technical information on characteristics, operation, costs analysis and human resources affected to LTP.

Questionnaire 1 was sent to all 28 ME that managed landfills with LTP. The one exception referred to a ME that manages two MSW landfills where leachate discharge is done directly to the local wastewater collection, for complete treatment at PWTP. With this questionnaire, a 93% (26 ME) response rate was obtained. The response rate obtained for Questionnaire 2 was 89% (i.e. 25 ME).

For the analysis phase, technical visits to all ME (i.e. 29) and LTP (i.e. 32) were also defined. These visits intended to verify actual LTP operational conditions, as well as to collect other pertinent information from landfill and LTP operators and managers. The information acquired in these visits regarded operational problems and conditions with leachate treatment and possible measures/modifications or reconversion plans previewed for the future. Another purpose for these visits was to clarify questions related with the questionnaire survey.

Taking in account the existing ME, leachates characteristics and treatment, previous studies, as well as current legal aspects landfills management, specially on leachate control and monitoring, 92 variables were selected and organised along with the questions of the survey (questionnaires 1 and 2). With these variables, 20 context information indicators and 69 performance indicators were considered.

All performance indicators, except opinion indicators, translate in general, variable ratios and refer to a one-year period (e.g. 2006).

4. Results and discussion

Main results obtained with methodology developed to apply performance indicators to leachate management in Portugal are presented in the according to the groups defined in section 3.1.

4.1 Context information

Context information was valuable to assess ME characteristics, existing disposal facilities, and LPT. Of the 29 ME, 26 reported 282 landfill infrastructures existing in their management area, with the distribution presented in Figure 2.

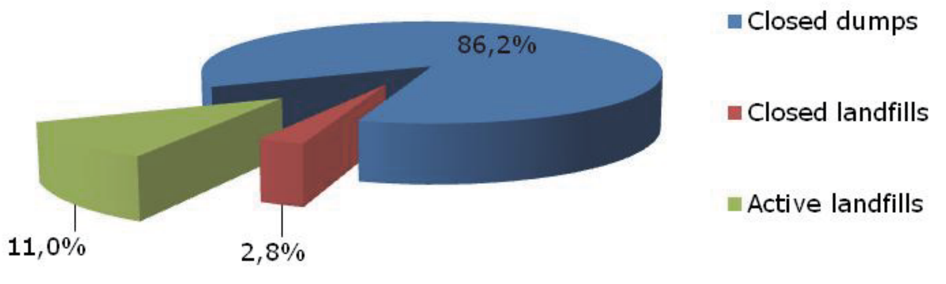

Fig. 2. Distribution of the number of disposal facilities reports by the ME

In terms of area and volume of the infrastructures where leachates produced are treated in LPT, results vary between 1.9 ha and 20.8 ha, or 190 906 m^3 to 2 501 022 m^3 of landfilled waste, which include old dumps, closed landfills and operational landfills.

The age of the 31 reported landfills (e.g. one closed landfill and 30 in operation), with the year 2006 in reference, varies between three and 10 years, or an average age of 6 years.

Most landfills (56%) were more than five years old and 34% were operational for seven years.
LTP leachate treatment processes are presented in Table 1. LTP treatments that include reverse osmosis membranes as final treatment for effluent discharge in streams represent 31% of the LTP. In two LTP (3%), macrophyte beds are used as final process. Eight LTP (25%) have treatment systems composed of activated sludge followed by physical and chemical treatments for partial treatment on-site, with following final off-site treatment at PWTP.

Treatment	Number of LTP	%
Aerated lagoon + reverse osmosis	5	15.6
Regularisation lagoon + reverse osmosis	1	3.1
Activated sludge + reverse osmosis	4	12.5
Physical and chemical treatment + evaporation + condensation + activated sludge	1	3.1
Stabilisation lagoons + aerated lagoon + macrophyte beds	1	3.1
Aerated lagoons	3	9.4
Stabilisation lagoons	1	3.1
Aerated lagoon + physical and chemical treatment	1	3.1
Aerated lagoon + macrophyte beds	1	3.1
Activated sludge	1	3.1
Activated sludge + physical and chemical treatment	8	25.0
Physical and chemical treatment + activated sludge	2	6.3
Physical and chemical treatment + Filter + activated sludge	1	3.1
Filter + activated sludge + physical and chemical treatment	1	3.1
Aerated lagoon + Filter + physical and chemical treatment	1	3.1
Total	32	100.0

Table 1. Distribution of LTP treatment processes

Of the 35 MSW landfills in operation plus one closed landfill that has a LTP, only four (11%), discharge generated leachates directly to wastewater collection systems for complete treatment at PWTP (Figure 3). The remaining 32 landfills have LTP, either to perform partial treatment for discharge to off-site treatment facility (i.e. 44% or 16 of 36 landfills) or complete treatment on site for further release to local stream (i.e. 42% or 15 of 36 landfills), existing one landfill where leachate discharge is null.
Of the 32 LTP, 19 were operating with no modifications previewed, eight had modifications previewed and five were deactivated for treatment system's modification.
In terms of LTP inclusion in the landfill original project, of the 30 LTP reported, 84% were designed and included in the landfill's design phase. In the remaining cases, landfills were designed and constructed before 1998, when it was still not legally determined LPT integration in the landfill design phase. Only 65% of the LPT were constructed according to original design. The main reasons pointed by the ME refer to the inability of the design treatment systems to comply with legal discharge limits, which resulted in LTP reconstruction.

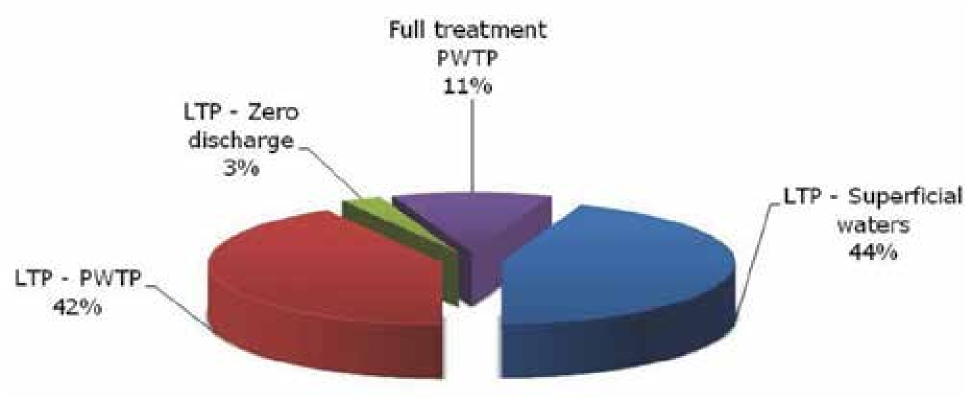

Fig. 3. Distribution of treatment systems and leachate discharge of MSW landfills

4.2 Environmental performance indicators

Leachate destination of the 282 landfilling infrastructures reported by the ME was only given for 26% of the cases. Of the total 282, final destination of leachates produced at 204 old dumps was not given (74%), 15% of the remaining infrastructures discharged produced leachates for partial or full treatment at LTP and 11% discharged leachates directly to sanitary sewers for full treatment at PWTP.

Of the remaining 38 of the 243 old dumps reported, nine discharged produced leachates in nearby landfill's LTP and 29 directly to sanitary sewers. About the eight closed landfills reported, four discharged generated leachates to nearby landfill's LPT and four to local sanitary sewers. Regarding the existing 35 landfills in operation, 89% (31) had LTP on-site either for partial or full treatment and the remaining 11% (4) discharged produced leachates directly to sanitary sewers.

Other environmental indicators refer to control and monitoring conformity with a number of legal demands, defined either in the landfill national legislation or in the landfill's environmental license, when existed. In terms of leachate control, according to the data retrieved from the ME, 61% of the 30 landfills reported did not show any unconformity. About groundwater control, 45% of the landfills revealed unconformities in the parameters that should be monitored. Considering superficial water control only 3% of the landfills revealed unconformities.

In terms of sludge produced by the leachate treatment systems, final destination reported was landfilling after dehydration processes. In the case of reverse osmosis membrane processes, concentrates are mainly recirculated to the landfill, according to the ME.

As indication of volume of leachate produced as a fraction of annual precipitation, results obtained for landfills reported by the ME were on average 41%. Only in the case of seven landfills, the values obtained were within the literature values of 15% and 50% (Ehrig, 1998).

In addition, relevant indicators for leachate production estimations were the results obtained for leachate production by area and by volume of landfilled waste (Figure 4 and Figure 5). The averages obtained were 3800 m^3 of leachate produced per hectare of landfilled waste per year and 0.039 m^3 of leachate per cubic meters of landfilled waste per year. These results exceed literature values. Bicudo e Pinheiro (1994) referred 2000 $m^3/(ha.year)$ and MacDougall et al. (2001) referred 0.005 $m^3/(m^3.year)$.

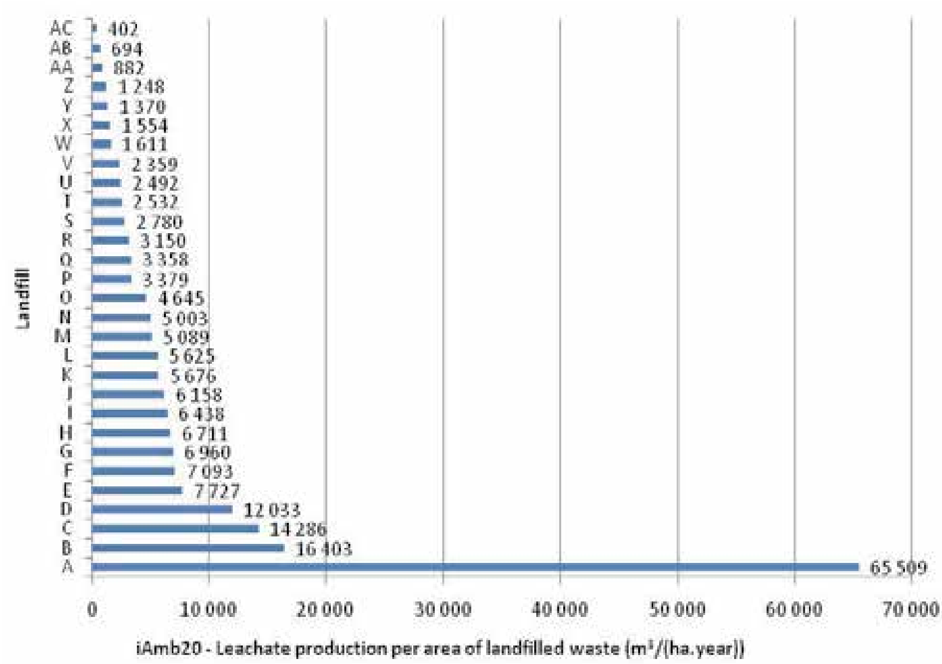

Fig. 4. Leachate production per area of landfilled waste per year for reported MSW landfills

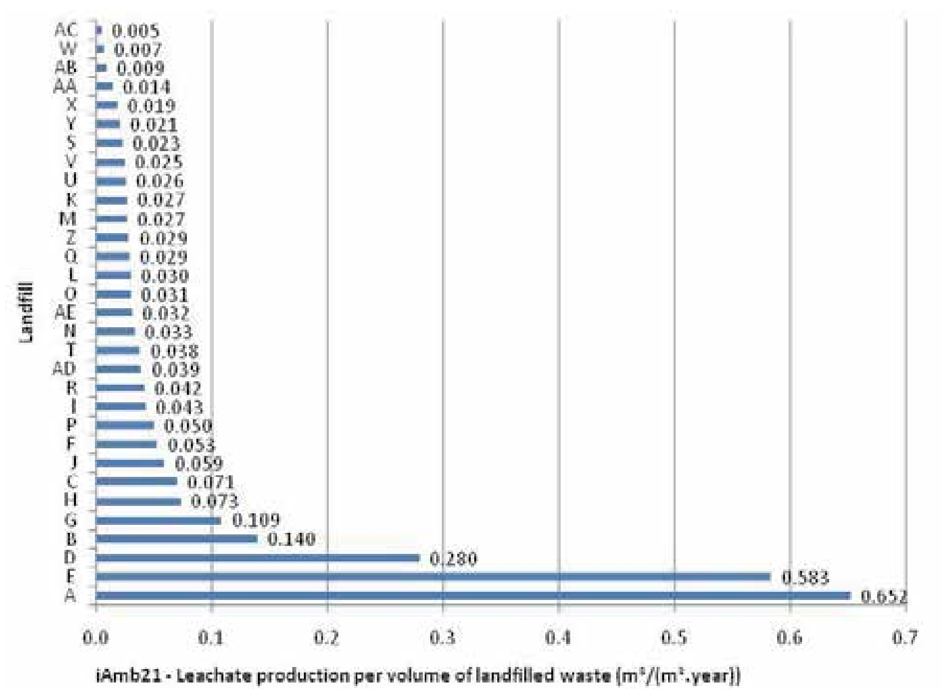

Fig. 5. Leachate production per volume of landfilled waste per year for reported MSW landfills

4.3 Human resources performance indicators

Human resources indicators determined, reveal that the 22 LPT where information was reported, 60% only have one operator, 31% two operators, and 9% three operators. It should be noticed that the number of operators reported by the ME in general do not account for the superior technician responsible for the LTP management. It is also noticed that in the case of small LTP operators are not entirely affected to LTP operation.

Concerning specific learning on LTP operation, only five cases referred conducting annually learning actions on LTP, mainly where reverse osmosis processes are used and in the case of the evaporation condensation treatment system.

4.4 Operational performance indicators

About problems identified on LTP functioning, ME reported in general operational and logistics problems and in a lesser extent personnel and other problems (Table 2). The operational problems identified were in general equipment damages, leachate storage capacity limitations, raw leachate quality treatability, as well as, in the case of reverse osmosis membrane reactors, high maintenance needs. Of the 23 ME that reported these problems, 40% indicated a monthly frequency and 32% a weekly frequency. In terms of logistic problems, eight ME reported mainly reagents supplies problems, three of them with a monthly frequency, other three rarely (i.e. once a year) and one with a daily frequency. Four ME, one with an annual frequency and three on a weekly basis reported personnel problems. The mentioned problems refer to lack of specialized personnel for the treatment system's operation. Five ME also mentioned other problems with a monthly frequency, however not specifically defined.

<table>
<tr><th>Problems</th><th>Operational</th><th>Logistics</th><th>Personnel</th><th>Other</th></tr>
<tr><td rowspan="4">Type</td><td>Equipment damages</td><td rowspan="4">Reagents supplies</td><td rowspan="4">Lack of specialized personnel</td><td rowspan="4">Not specified</td></tr>
<tr><td>Leachate storage capacity limitations</td></tr>
<tr><td>Reverse osmosis membrane reactors, high maintenance needs</td></tr>
<tr><td>Raw leachate quality treatability</td></tr>
<tr><td>Frequency of occurrence</td><td>23 reported:
-13% weekly
-32% monthly
-40% per trimester
-13% yearly</td><td>8 reported:
-1 weekly
-1 monthly
-3 per trimester
-3 yearly</td><td>4 reported:
-1 weekly
-3 yearly</td><td>5 reported:
-5 weekly</td></tr>
</table>

Table 2. Problem types and frequency of occurrence at reported LTP

Regarding leachate and groundwater monitoring and according to the information given by the ME in the questionnaires of 27 landfills, in 21 (78%) 100% of the number of leachate parameter analysis defined in the legislation or in the landfill environmental license were done. Five landfills performed between 80% and 99% of the total number of analysis. As for groundwater monitoring where information was given, 54% (i.e. 13 of 24 landfills)

performed all parameter analyses legally defined, seven landfills between 80% and 99%, and the remaining four landfills below 79% of the number of groundwater parameter analysis.
LTP energy consumption was also determined and an annual average of 11.1 kWh/m^3 of leachate was obtained, with values varying between 1.8 kWh/m^3 and 38.0 kWh/m^3.

4.5 Financial and economic performance indicators

Concerning LTP cost analysis, the performance indicators attempted to translate LTP overall costs. Results are based on the information reported in the questionnaires, however ME only reported this information for 17 LTP, lacking information on few cost components in some cases. On the other hand, the values obtained are relevant for reference and comparison between the LTP treatment systems.
Average overall unit costs (i.e. per unit of raw leachate treated in LTP) for the year 2006 was 8.8 €/m^3, 6.1 €/m^3 referring to current expenses costs and 2.7 €/m^3 to capital costs (i.e. capital amortizations in 2006). In terms of main treatment systems, treatments that use macrophyte beds revealed to be the less expensive (2.4 €/m^3). The evaporation /condensation process, recently being used in one LTP, presented the highest capital costs (25.0 €/m^3). The ME did not report in this case current expenses costs and total unit costs could not be determined. Other treatments refer to all remaining treatments systems presented in Table 1. Except for the evaporation/condensation treatment system, the average unit cost for these treatments is the higher obtained (8.5 €/m^3), mainly due to one of the LTP that presented higher costs comparing with other LTP with similar treatment systems (i.e. in terms of treatment system reconstruction costs and current expenses costs), thus increasing the unit cost. Comparing with other treatments systems the reverse osmosis membrane process presented on average higher capital costs (3.3 €/m^3).
Percentage distribution of current expenses costs obtained (Figure 6) revealed that on average 67% refer to other current expenses costs (e.g. reagents, equipment rental, service acquisitions and other costs), 23% refer to energy costs for LTP operation, and the remaining 10% to personnel costs.

4.6 Service quality performance indicators

The main leachate contaminants (BOD_5, COD, total nitrogen and TSS) removal efficiencies were determined for 21 LTP. Taking in account the information on raw leachate and treated leachate quality monthly information for 2006, reported in the questionnaires by the ME, Table 3 presents removal efficiencies obtained for the main treatment systems.
As previously presented, treatment systems with macrophyte beds are less expensive, although the removal efficiencies are rather low (Table 3). In the case of total suspended solids, no removal was obtained. Considering the discharge to sanitary sewers this treatment option can be economic. The reverse osmosis membrane process revealed to be the most contaminant removal efficient treatment option as it is mainly used when discharge to streams is the only option. Although only COD removal efficiency was possible to determine for the evaporation/condensation process, it also shows to be a possible option, however expensive, for full treatment on-site and discharge to streams. The remaining treatments systems of nine LTP showed various removal efficiencies for the considered parameters. These treatment processes are mainly used for partial treatment on-site, and further complete treatment at PWTP. With respect to pH, all LTP effluents complied with legal limit values (i.e. pH between 6 and 9) for discharge to stream.

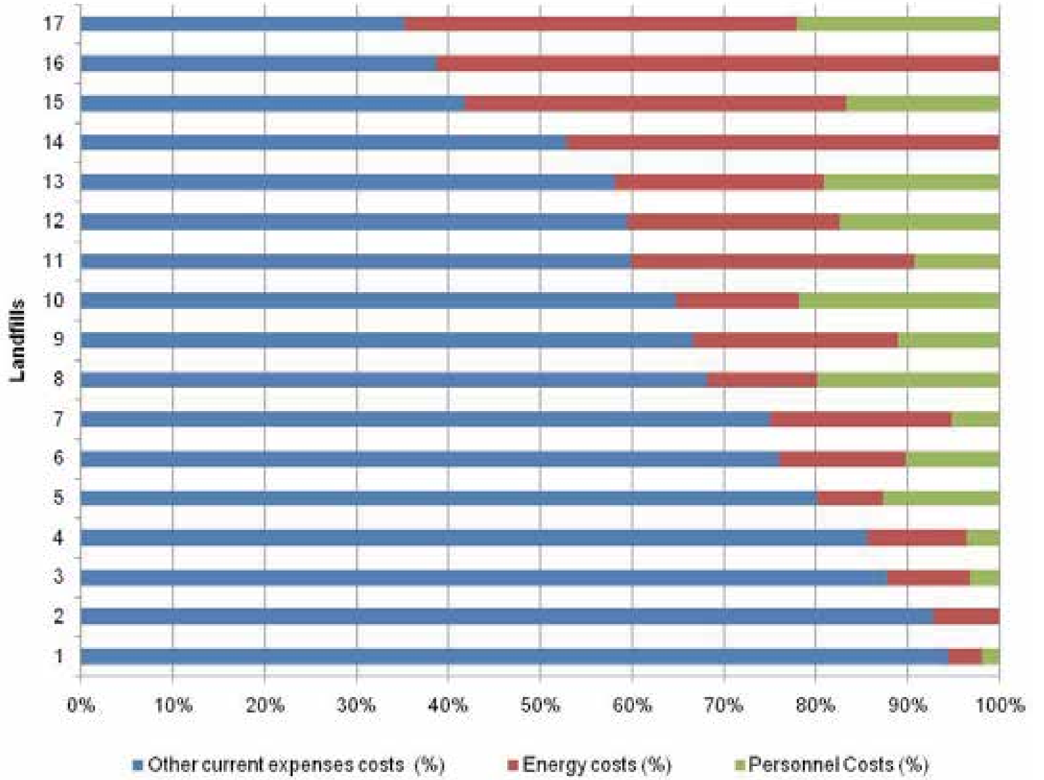

Fig. 6. Percentage distribution of current expenses costs for reported MSW landfills

Main leachate treatments	Number of LTP	Removal efficiency (%)								
		COD			Total Nitrogen			TSS		
		Min	Max	Average	Min	Max	Average	Min	Max	Average
Macrophyte beds	2	26.6	49.3	37.9	17.4	17.4	17.4	No removal		
Reverse osmosis	9	98.6	99.9	99.6	99.3	99.8	99.6	87.9	99.5	93.7
Evaporation/Condensation	1	99,9			Not available			Not available		
Other treatments	9	53.0	89.6	69.0	29.0	46.6	37.8	18.8	94.9	54.2

Table 3. Average, minimum, and maximum leachate contaminant removal efficiencies for the main treatment systems

4.7 Opinion indicators

This group of indicators pretended to transmit the questionnaires' respondent, in general LTP or landfill managers, about LTP performance. Results are presented in Figure 7. In the case of adequacy of the treatment system to leachate quantity, 48% of the respondents positioned in the middle (i.e. nor satisfied, nor unsatisfied). Similar percentage of responses

(26%) was obtained both for the positive pole (i.e. satisfied or very satisfied) and for the negative pole (i.e. unsatisfied or very unsatisfied). In terms of leachate quality, 60% of the responses were in the middle position, although 29% were negative, revealing that managers are more concerned about leachate quantity than quantity on the adequacy of the leachate treatment systems.

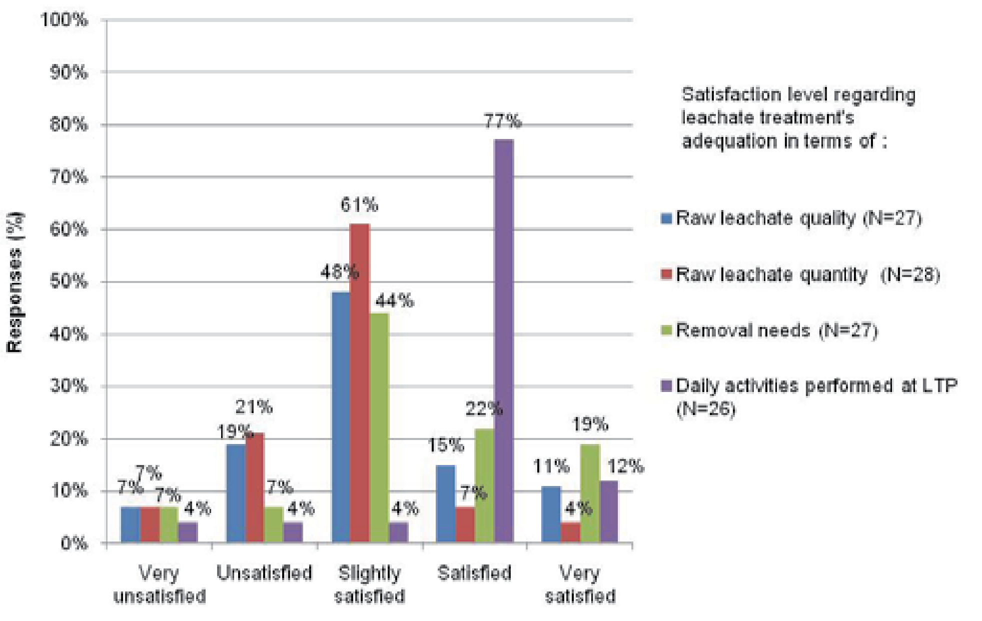

Fig. 7. Opinion indicators results

5. Conclusion

Performance indicators and relevant context information can be a valuable tool on MSW landfills leachate management assessment and benchmarking analysis. With the application of the proposed performance indicators to the leachate treatment and management in Portugal's mainland it was possible to identify the most cost and contaminant removal efficient treatments systems, among several constrains regarding the lack of specific definitions on leachate discharge quality limits to streams and lakes, considering the particular characteristics of this effluent. To discharge in sanitary systems, more economic treatments can be used, however legal definition and uniformity regarding discharge quality limits in domestic wastewater collection systems is also needed. In the case of old dumps, the monitoring and management is generally defined on national legislation. Therefore, a need for management definition and for leachate monitoring parameters generated by closed dumps would be an improvement in this matter.

On the other hand, most problems identified possibly relate to an inadaptability of general leachate production and quality models with the national specific meteorological and landfill operation conditions. On this matter, an historical assessment on MSW landfills could be developed to adapt existing models to the Portuguese context. Regarding leachate and concentrate recirculation on current operational MSW landfills, further studies to assess economic and environmental costs and benefits should also be developed. In this way, legal authorities could have relevant information for decision making in modifying existing legislation on this matter.

6. Acknowledgment

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Measurements of Carbonaceous Aerosols Using Semi-Continuous Thermal-Optical Method

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1. Introduction

Waste management involves collection, transport, processing, recycling, disposal, and monitoring of waste materials that can be solid, liquid, gaseous, or radioactive, which all are generated by human. It is important to monitor aerosols emitted during waste treatment and management to understand their impact on human health and the environment. Carbonaceous aerosols are major components in air pollution as a result of energy consumption, thus measurement of them is important to waste management. Increasing interest has been drawn to the identification, measurement, analysis, and modeling of carbon aerosols in the past decade. This book chapter will provide a review of the widely used semi-continuous thermal-optical method to determine carbonaceous aerosols in relation to air pollution and waste management.

Quantification of carbonaceous species provides important observations in understanding aerosol life cycle. Carbonaceous aerosols play important roles in air quality, human health, and global climate change. However, accurate measurement of carbonaceous particles still presents challenges. Carbonaceous particles are divided into three categories: organic carbon (OC), elemental carbon (EC), and inorganic carbonate carbon (CC) [Chow et al., 2005; Schauer et al., 2003]. The terms "elemental carbon (EC)", "soot", "black carbon", "graphic carbon", and "light absorbing carbon" are often used loosely and interchangeably in different research areas. Atmospheric EC particles are produced almost exclusively under incomplete combustion conditions. They are from both anthropogenic and biogenic emissions. Ambient elemental carbon particles rarely appear as diamond crystalline structure. EC aerosols absorb light effectively and they can be characterized by light scattering, absorption, or transmittance, as well as other methods. Absorption spectroscopy is deemed to provide quantitative information of EC. Difference in the definition of EC is a result of measurement methods [Jeong et al., 2004; Watson et al., 2008].

Increasingly OC has drawn more attention because of its effect on regional air pollution and global climate change. OC aerosol formation is attributed to both biogenic and anthropogenic sources [Bond & Bergstrom, 2006]. OC may be released directly into the atmosphere (primary organic aerosol) or formed when gaseous volatile organic compounds are released to the atmosphere followed by photolysis induced oxidation to form secondary organic aerosols [Bae et al., 2004; Schauer et al., 2003]. Past findings indicate that a large percentage of OC observed around the world is secondary [Zhang et al., 2007]. This chapter, however, focuses on the widely used semi-continuous thermal analysis method. Comparisons among relevant methods are also provided.

2. Thermal desorption analysis methods

Thermal desorption has been used to analyze volatile organic compounds. The physical principle lies in the fact that different components of a sample volatize, oxidize, or react with other reagents as the temperature profile changes [MacKenzle, 1970]. Many methods employ a two-step temperature profile. Generally speaking, sample is heated in the first step to a temperature ranging from 350 °C to 850 °C. Carbon evolved in this step is defined as OC. In the second step, sample is heated to a temperature ranging from 650 °C to 1100 °C. Carbon evolved in this step is defined as EC. At the first temperature regime, the volatilization rate of EC is assumed to be low, and OC evolution occurs in an atmosphere without an oxidizing agent. Carbon dioxide (CO_2) gas forms as a result of OC evolving from the sample. In step 2, an oxidizer is introduced. Oxygen (O_2) is often used. EC reacts with this oxidizing agent, sometimes under catalysis conditions, to form CO_2. CO_2 is detected directly. A methane (CH_4) – helium (He) mixture is used to calibrate the system; the CH_4 is oxidized in the same manner to achieve quantification. The original compounds are transformed due to thermally-induced reactions (dissociation or oxidation). The detection is not chemically specific using the thermal analysis method. Results are often reported as empirically and operationally defined categories including OC, EC, and TC. TC is the sum of OC and EC (TC=OC+EC).

An important factor in thermal evolution methods is the OC/EC split point. Many methods use Optical Reflectance and/or Optical Transmission to monitor the conversion of OC to EC and the oxidation of EC to CO_2. The rationale is that since EC is not volatile until very high temperatures (well above the ~840 °C used by the NIOSH method, for example), its release is only dependent on oxidation when oxygen is present. High temperatures in the non-oxidizing environment often cause some OC components to form EC by charring. This complicates the determination of EC as additional EC is formed due to this charring. When oxygen is added to the sample oven, the black EC char will combust and the filter becomes white. When the light intensity from reflection or transmission of the samples on the filter reaches its original intensity, the charred OC is assumed to be removed. The OC/EC split point is usually defined in this manner. It is assumed what comes off after the split point is quantitatively nearly equal to the EC that was on the filter originally as EC.

Thermal-Optical methods assume that: (1) The EC caused by charring of OC's during the first O_2-free step is more easily oxidized; or (2) that the absorption coefficient of the EC formed by charring is similar to the absorption coefficient of the original EC within the filter. If either of these assumptions is correct, then the method will be an effective quantitative method of OC and EC. Although the operational principle is similar, subtle differences exist among the different methods. These factors may include analysis atmosphere, temperature profiles, optical monitoring approaches, sample size, and other differences in physical configurations of the analytical instrument [Watson et al., 2005; Chow et al., 2005]. Some examples of more detailed studies of the effect of using TOT and TOR on the OCEC split point are discussed elsewhere [Chow et al., 2004; Cheng et al., 2009].

Particulate samples are usually collected using filters ranging from several hrs to days, then samples are prepared for off-line analysis in the laboratory. For OC and EC laboratory analysis, the Sunset instrument (Sunset Laboratories Inc.) and the DRI (Desert Research Institute) instrument are among the most commonly used. Near real-time or real-time on-line techniques are advantageous compared with off-line ones, because they provide faster sampling resolution and reduce labor in analysis. More importantly, the faster time resolution makes it possible to capture fast changing fluctuations of particle emisisons, where the off-line methods would have missed due to the longer sampling time.

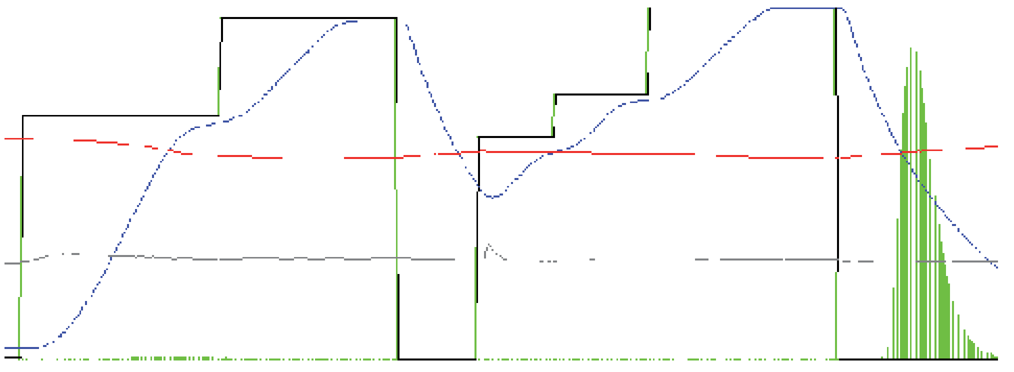

Fig. 1. An example of the modified NIOSH thermo-optical analysis thermal desorption diagram of a field sample. The x-axis is time in seconds, and y-axis is intensity of different traces. The blue color is oven temperature; red NDIR laser intensity; gray pressure; and green carbon dioxide.

Several techniques are established for in situ determination of black carbon (BC), such as the aethalometer and the particle soot absorption photometer. The relationship between BC and EC, however, is not fully resolved. These on-line EC methods do not provide OC measurements simultaneously. The Sunset Semi-Continuous Organic Carbon/Elemental Carbon (OCEC) Aerosol Analyzer has been a successful development for on-line OC and EC measurement. It can provide measurements of OC and EC on hourly time scales, and it allows for semi-continuous sampling with analysis immediately after sample collection. The instrument provides quantification of both OC and EC aerosols and requires no off-line sample treatment and laboratory analysis. This reduction in complexity, along with the ability to measure OC and EC on an hourly basis, provides advantages over conventional off-line integrated techniques.

Aerosol light absorption can be used to determine EC (or BC) either on filter media or in situ. There are several commerically avaialble instruments based on aerosol light absorption including the aethalometer, particle soot absorption photometer (PSAP), micro soot sensor, multi-angle absorption photometer (MAAP), photo-acoustic soot spectrometer (PASS), and single particle soot photometer (SP2). Moosmüller et al. [2009] provides a detailed review of these techniques. Due to the commericial avaiability of these fast in situ instruments, more comparisons have been made to the EC measurements among them. Instrument uncertainty and minimum detection limits were determined for these techniques. Some recent examples of these quantities and comparisons are seen in Chow et al. [2009], Cross et al. [2010], Slowick et al. [2007].

Other newer developments often involve mass spectrometery. One such successful example is the aerosol mass spectrometer [Jayne et al., 2000]. However, it does not provide simultaneous EC measurements, although it can provide faster resolution of total organic aerosol. The latter is often deduced to primary and secondary components using positive matrix factorization (PMF) analysis. As a result, it is more labor intensive to operate and conduct data reduction. In addition, MS based instruments are often more expensive to purchase. They take more power and space, therefore, not immediately accessible for long-term regulatory monitoring purpose in waste management.

2.1 The Sunset OCEC analyzer

The semi-continuous Sunset OCEC analyzers (Model 3F, Sunset Laboratory Inc., Portland, OR) is widely used to measure OC and EC mass loadings at different locations. Ambient samples were collected continuously by drawing a sample flow of ~8 lpm. A cyclone was used upstream of the instruments to pass particles smaller than 2.5 µm. The airstream also passed through a denuder to remove any volatile organic compounds in the air. Sample flow rate was adjusted for the pressure difference between sea level and each of the sites to ensure accurate conversion of sample volume. During automated semi-continuous sampling, particulate matter was deposited on a quartz filter. The quartz filter was normally installed with a second backup filter, mostly to serve as support for the front filter. The portion of the sample tube containing the quartz filter was positioned within the central part of an oven, whose temperature was controlled by an instrument control and data logging program installed on a laptop computer and interfaced with the OCEC instrument.

After a sample was collected, in situ analysis was conducted by using the modified NIOSH method 5040, i.e., thermal optical transmittance analysis, to quantify OC and EC. The oven was first purged with helium after a sample was collected. The temperature inside the oven was ramped up in a step fashion to ~ 870 °C to thermally desorb the organic compounds. The pyrolysis products were converted to carbon dioxide (CO_2) by a redox reaction with manganese dioxide. The CO_2 was quantified using a self-contained non-dispersive infrared (NDIR) laser detection system. In order to quantify EC using the thermal method, a second temperature ramp was applied while purging the oven with a mixture containing oxygen and helium. During this stage, the elemental carbon was oxidized and the resulting CO_2 was detected by the NDIR detection system. At the end of each analysis, a fixed volume of external standard containing methane (CH_4) was injected and thus a known carbon mass could be derived. The external calibration was used in each analysis to insure repeatable quantification. The modified NIOSH thermal-optical transmittance protocol used during a field study in Mexico City is summarized in Table 1.

Errors induced by pyrolysis of OC are corrected by continuously monitoring the absorbance of a tunable diode laser beam (λ = 660 nm) passing through the sample filter. When the laser absorbance reaches the background level before the initial temperature ramping, the split point between OC and EC can be determined. OC and EC determined in this manner are defined as Thermal OC and Thermal EC. Total carbon (TC) is the sum of Thermal OC and Thermal EC, TC = Thermal OC + Thermal EC, or TC=OC+EC. The Sunset OCEC analyzer also provides an optical measurement of EC by laser transmission, i.e. Optical EC. Optical OC can be derived by subtracting Optical EC from total carbon, Optical OC = TC - Optical EC, where TC is determined in the thermal analysis.

Modifications can be made to the temperature steps in the thermal-optical method. Conny et al. [2003] conducted a study to optimize the thermal-optical method for measuring atmospheric black carbon employing surface response modeling of EC/TC, maximum laser attenuation in He, and laser attenuation at the end of the He phase. They tried to minimize the positive bias from the detection of residual OC on the filter as native EC by maximizing the production OC char by the Sunset (TOT) instrument. In addition, they sought to minimize the negative bias from the loss of native EC at high temperatures. This first study concluded that for particle samples around 30 to 50 μg, the optimal condition for steps 1- 4 in the He environment are 190 ºC for 60 s, 365 ºC for 60 s, 610 °C for 60 s, and 835 °C for 72 s, respectively.

Carrier Gas	Duration (sec)	Temperature (ºC)
He-1	10	Ambient
He-2	80	600
He-3	90	870
He-4	25	No Heat
O_2-1	30	600
O_2-2	30	700
O_2-3	35	760
O_2-4	105	870
CalGas	110	No Heat

Table 1. An example of the modified NIOSH 5040 thermal-optical protocol used during the MILAGRO campaign [Yu et al., 2009].

Recently, Conny et al. [2009] reported an update using the same empirical factorial-based response-surface modeling approach to optimize the thermal-optical transmission analysis of atmospheric black carbon. They showed that the temperature protocol in the TOT analysis of a Sunset Instrument can be modified to distinguish pyrolyzed OC from BC based on the Beer-Lambert Law. The optimal TOT step-4 condition in the helium environment was established to be around 830 - 850 °C using urban samples via response surface modeling in their newer findings, although temperature as low as 750 °C or as high as 890 °C is not excluded. This optimization is based on two criteria. First, sufficient pyrolysis of OC must occur in the high temperature helium environment (i.e., He step 4 or the high temperature step in He), so that insufficiently pyrolyzed OC is not measured as native BC after the split point. Second, the apparent specific absorption cross sections of OC char and the apparent specific absorption cross sections of native BC determined by the instrument are assumed to be equivalent to determine the optimal operation conditions.

2.2 Aerosol sampling inlet and field deployment

In order to eliminate interference from near ground activities, an aerosol sampling stack can be used adjacent to the dwelling hosting the instrument at a surface site. An example is given below based on our field deployment experience. The sampling stack is made of PVC pipe ~ 20 cm in diameter and extending ~ 8 m above ground. The stack inlet is protected by a rain cap. A heated stainless steel sampling intake tube (~ 5 cm in diameter) is coaxially positioned in the center of stack ~ 4 m below the top of the stack and extending through the lower end cap. The airflow through the aerosol sampling stack is ~ 1000 lpm, of which approximately 120 lpm is drawn into the heated tube. The tube is wrapped with heating tape and insulation and further encased in a PVC pipe. Electric power is applied to heat the

sample line such that the relative humidity (RH) of the sample air is maintained at or below 40%. Much simpler design can be used to obtain equally good sampling results.
Filters are recommended to be changed every few days before the laser correction factor reached below ~ 90%. Sampling interval shall be determined based upon local mass loadings. At locations with low mass loadings that are close to the instrument detection limits, it makes sense to sample for longer time. Otherwise, for semi-real time sampling, the sample time is usually chosen to be one hour, i.e., 45-minute ambient sampling followed by 15 minutes thermal-optical analysis. Daily, at midnight, a 0-min sampling blank is taken. Instruments should be calibrated using an external filter with known OC and EC mass concentrations. Values reported are corrected to ambient temperature and pressure, this is especially important if the sampling location is elevated. Externally produced standard filters are recommended to check the precision of instrument as additional quality assurance. The relative standard deviations deduced from collocated in situ measurements between the two analyzers are determined to be 5.3%, 5.6%, 9.6%, and 4.9% for Thermal OC, Optical OC, Optical EC, and TC, respectively [Bauer et al., 2009]. The limits of detection for OC and EC determined using the thermal-optical method by the Sunset instrument were estimated to be approximately 0.2 μgC/m^3 [Schauer et al., 2003]. Readers are referred to previous reviews to find more details about differences among major instruments for determination of particulate carbonaceous compositions [Chow et al., 2007].

2.3 Thermal carbon and optical carbon

Optical vs. Thermal	Slope	R^2	Locations	Reference
OC	0.93±0.01	0.95	Mexico City T1	[Yu et al., 2009]
	0.84±0.02	0.37	Mexico City T2	[Yu et al., 2009]
EC	0.89±0.02	0.95	Rochester, NY	[Jeong et al., 2004]
	0.99±0.07	0.73	Philadelphia, PA	[Jeong et al., 2004]
	0.58±0.05	-*	New York City	[Venkatachari et al., 2006]
	1.03**	0.94	Mt. Tai, China	[Kanaya et al., 2006]
	0.91	0.84	3 sites in New York & 1 site in Turkey	[Ahmed et al., 2009]
	1.43±0.01	0.96	Mexico City T1	[Yu et al., 2009]
	1.39±0.01	0.91	Mexico City T2	[Yu et al., 2009]

* Not available from the original reference
** Derived from the slope of the linear least-squares analysis of thermal EC vs. optical EC

Table 2. Linear least-squares fit parameters between quantities determined using optical and thermal-optical approaches

The thermally determined quantities are considered reliable and are used for data reporting. Some recent studies have looked into the correlation between the thermal-optically determined quantities thermal OC and thermal EC, and shown that these quantities may be strongly correlated (Table 2). Strong linear relationships have been seen at multiple locations with reasonable R^2. However, the values of the fitting slope vary from ~ 0.6 to ~ 1.4. This indicates that no single simple numerical relationship can be applied everywhere. One also needs to take into consideration that some of these studies were conducted at locations of low EC mass loadings, which contributes to higher uncertainty in the analysis results. In the future, similar studies should be done at locations of higher carbonaceous mass loadings, which would make such comparisons more conclusive. More studies have compared the EC quantities determined by different in situ techniques. It is still an on-going effort to determine the differences among these methods [Chow et al., 2009; Cross et al., 2010; Slowick et al., 2007].

2.4 Carbon monitoring at different locations

Carbonaceous aerosols have been monitored by established networks in the U.S. such as the Interagency Monitoring of Protected Visual Environments (IMPROVE) and the Speciated Trends Network (STN). Many intensive field studies have been conducted to study carbonaceous aerosols in U.S. in addition to the monitoring by the long-term network. No strong correlations have been seen among OC and other major particulate matter components such as sulfate, nitrate, or ammonium ions based on a recent study compiling available ground-based carbon data worldwide [Bahadur et al., 2009]. As more attention has been directed to the importance of carbonaceous aerosols, more field data would become available.

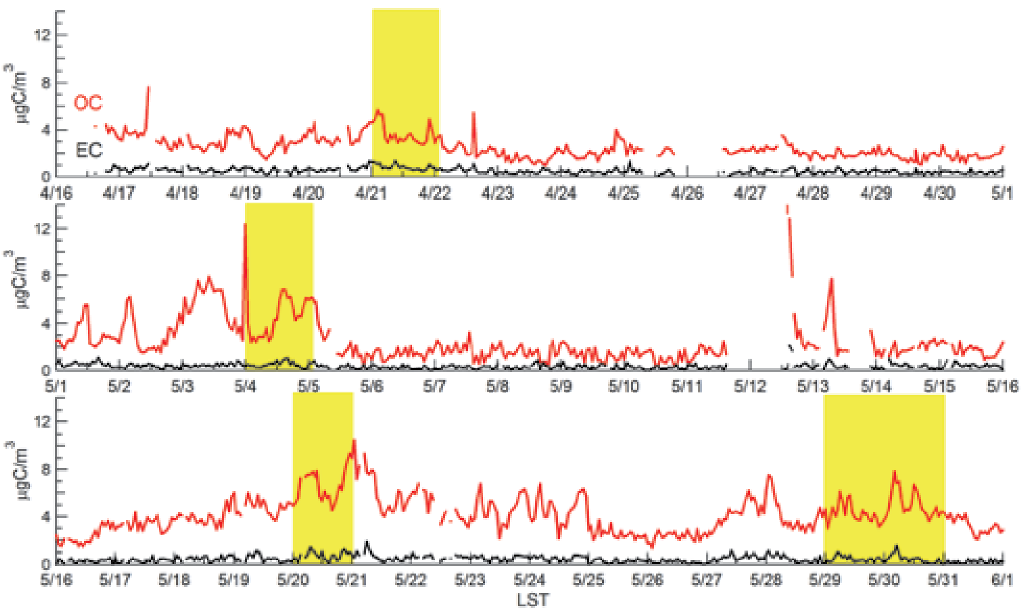

Fig. 2. Time series of organic carbon (OC) and elemental carbon (EC) measured at an urban site in Houston, TX in 2009. The yellow highlighted area indicates local ozone observation was over 75 ppb.

Table 3 shows a comparison of $PM_{2.5}$ OC and EC with other metropolitan areas in the world, such as Beijing, Shanghai, Hong Kong, Los Angeles, and Houston. Most of these OC and EC measurements were obtained by thermal optical reflectance methods [Birch, 1998; Cachier et al., 1989; Chow et al., 2001]. Since the definitions of OC and EC are operationally defined, uncertainties exist among different methods. The OC:EC values for T1 and T2 reported in Table 3 are obtained by Deming regression analysis. The OC:EC value obtained at T1 is

Location	OC:EC	OC avg	EC avg	TC	Season	Method	Reference
		μgC/m³					
Beijing	2.4	9.4	4.3	--	Summer	Rupprecht ambient carbon particulate monitor	[Yu et al., 2006]
Beijing	3.0	20.4	6.6	26.9	Fall	Rupprecht ambient carbon particulate monitor	[Duan et al., 2005]
Shanghai	--	7.9	3.5	11.4	Summer	Sunset OCEC analyzer NIOSH protocol	[Feng et al., 2006]
Guangzhou	--	14.5	6.3	20.8	Summer	Sunset OCEC analyzer NIOSH protocol	[Feng et al., 2006]
Hong Kong	2-3	12	6	--	Winter	Thermal manganese dioxide oxidation	[Ho et al., 2002]
Hong Kong	2.4	14.7	6.1	--	Winter	IMPROVE thermal optical reflectance method	[Cao et al., 2003]
Houston	2.9-4.8	2.4-4.3	0.3-0.6	--	All	NIOSH thermal optical reflectance method	[Russell and Allen, 2004]
Los Angeles	2.5	8.3	2.4	2--	Summer	IMPROVE thermal optical reflectance method	[Chow et al., 1994]
Milan	4.2	5.2	1.2	--	Summer	NIOSH thermal optical reflectance method	[Lonati et al., 2007]
Madrid	2.7	4	1	--	Summer	EPA thermo-optical transmittance technique	[Plaza et al., 2006]
Barcelona	2.8	3.9	1.9	5.8	Summer	Sunset OCEC analyzer NIOSH protocol	[Viana et al., 2007]
Amsterdam	2.6	3.6	1.5	5.1	Summer	Sunset OCEC analyzer NIOSH protocol	[Viana et al., 2007]
US rural	2.3-4.0*	--	--	--	Summer	IMPROVE thermal optical reflectance method	[Schichtel et al., 2008]
US urban	1.1-1.7*	--	--	--	Summer	IMPROVE thermal optical reflectance method	[Schichtel et al., 2008]
Mexico	1.7**	9.9	5.8	15.8	Spring	IMPROVE thermal optical reflectance method	[Chow et al., 2002]
Mexico –T1	--	3.7	4.0	16	Spring	IMPROVE thermal optical reflectance method	[Querol et al., 2008]
Mexico – T1	--	5.0	1.6	--	Spring	Sunset OCEC analyzer modified NIOSH protocol	[Stone et al., 2008]
Mexico – T1	--	6.1	1.5	8.2	Spring	Sunset OCEC analyzer modified NIOSH protocol	[Hennigan et al., 2008]
Mexico - T1	0.9	6.4	2.1	8.5	Spring	Sunset OCEC analyzer modified NIOSH protocol	[Yu et al., 2009]
Mexico – T2	10.1	5.4	0.6	6.0	Spring	Sunset OCEC analyzer modified NIOSH protocol	[Yu et al., 2009]

* Derived from EC/TC 82nd-98th percentile ratios
**Derived from OC/TC
-- Not available from original references

Table 3. Comparison of $PM_{2.5}$ OC:EC, OC, EC, and TC observed in different cities

comparable to the average reported for urban US cities [Schichtel et al., 2008]. In contrast, the average OC:EC value at T2 is comparable to places such as Houston [Russell and Allen, 2004] and Milan [Lonati et al., 2007]. It is close to the average reported for US rural areas [Schichtel et al., 2008].

We also need to take into account the season when measurements were taken when comparing results from different locations. For example, winter observations usually result in higher mass loadings than those in summer, most likely affected by boundary layer height and mixing. For example, when looking into recent results from Mexico city, a more sensible comparison is with that in a study in Mexico in 1997 [Chow et al., 2002]. Six core sites were used in this study, La Merced, Pedregal, Xalostoc, Tlalnepantla, Netzahualcoyotl, and Cerro de la Estrella, mostly representing urban, suburban, residential, industrial, and commercial areas in or near downtown Mexico City. Results reported were averages of all six sites. The T1 and T2 comparisons with these results are in reasonable agreement. However, direct comparison with results from the regional sites may be more useful in illustrating changes or trends over the past decade. Unfortunately, the latter were not available. Querol et al. recently reported the OC and EC results during MILAGRO [Querol et al., 2008], but only results from T1 were available for comparison. Since Querol et al., [2008] selected only a few 6 hr samples to determine OC and EC, their results do not have the same time resolution or as many samples as reported here. We expect, therefore, that the results with higher time resolution may provide more complete statistics because of the continuous hourly measurements.

3. Data reduction

Although the values of OC:EC and EC:TC could be used to get some idea of the extent of primary and secondary organic carbon, quantification of POC and SOC is important to assess the performance of organic aerosol predictions made by models. Identification of POC and SOC is quite important in further analysis. Due to the lack of an analytical technique for directly quantifying the atmospheric concentrations of primary organic carbon (POC) and secondary organic carbon (SOC), indirect methods have been developed to estimate their concentrations. Here we will provide detailed description of the widely used semi-empirical EC tracer method, because it is simple to use.

3.1 The EC tracer method

The semi-empirical EC tracer method is used to derive POC and SOC empirically. The assumptions and methodology of EC tracer method are described in detail elsewhere [Castro et al., 1999; Turpin and Huntzicker, 1991; 1995; Yu et al., 2007]. Briefly, total OC (OC_{total}) is defined as the sum of POC and SOC, Eq. (1).

$$\mathrm{SOC} = \mathrm{OC}_{\mathrm{total}} - \mathrm{POC} \tag{1}$$

POC is defined in Eq. (2),

$$\mathrm{POC} = \mathrm{EC} \times \left(\frac{OC}{EC} \right)_{pri} \tag{2}$$

where $(OC:EC)_{pri}$ is the estimated primary carbon ratio. The OC emitted from non-combustion sources, such as emission directly from vegetation, is assumed to be negligible in the approach used here. Using the minimum OC to EC ratio, $(OC:EC)_{min}$, to substitute for $(OC:EC)_{pri}$, the SOC and POC can therefore be estimated [Cabada et al., 2004; Castro et al., 1999]:

$$\mathrm{SOC} = \mathrm{OC}_{\mathrm{total}} - \mathrm{EC} \times \left(\frac{OC}{EC}\right)_{\mathrm{min}} \tag{3}$$

Several assumptions must be made to deduce SOC and POC in this manner. For instance, samples used to calculate $(OC:EC)_{min}$ have negligible amounts of SOC. Composition and emission sources of POC and SOC are assumed to be relatively constant spatially and temporally. Contribution from non-combustion POC is assumed low. Contribution from semi-volatile organic compounds is also assumed to be low compared with non-volatile organic species. The determination of $(OC:EC)_{min}$ is crucial in this approach.

The EC tracer method is mainly dependent on ambient measurements of OC and EC and therefore is easy to use. The key is to estimate $(OC:EC)_{pri}$ from ambient conditions. The challenge lies determining $(OC:EC)_{pri}$, because it could be influenced by meteorological conditions and emission fluctuations [Turpin and Huntzicker, 1995; Yu, S. et al., 2004].

Previous authors often used the lowest 5% or 10% measured OC/EC values in a given season to estimate $(OC:EC)_{min}$ [Lim and Turpin, 2002; Yuan et al., 2006]. It is worth mentioning that Yuan *et al.* found that $(OC:EC)_{pri}$ is seasonally-dependent. For instance, the $(OC:EC)_{pri}$ ranged from 0.41 to 0.88 from summer to winter based on observations in Hong Kong [Yuan et al., 2006]. Therefore, the $(OC:EC)_{pri}$ determined in a particular study could not be used in all seasons elsewhere.

In addition, other approach can be used to obtain $(OC:EC)_{pri}$, since sometimes the R^2 values from the lowest 5% OC:EC approach may not be as satisfactory. For example, the linear least-squares fit results of OC vs. EC were grouped by binning OC:EC values in different ranges at the study site in Mexico City [Yu et al., 2009]. The $(OC:EC)_{min}$=0.61 at T1 falls in the range of OC:EC values typical of fossil fuel sources. The R^2 value obtained is 0.95. On the other hand, $(OC:EC)_{min}$ is 2.26 with the R^2= 0.86 at T2, a rural site in Mexico City. The $(OC:EC)_{min}$ value at T2 falls in the range of OC:EC values typical of biomass emissions [Gelencser et al., 2007]. The results from this approach are in reasonable agreement with those using the lowest 2.5% or 5% of OC:EC data. Since the results obtained by binning the OC and EC values to different ranges prior to applying linear least-squares analysis yields improved R^2, the slopes from this regression analysis may be used as $(OC:EC)_{min}$=$(OC:EC)_{pri}$ to derive SOC and POC.

The intercepts from the regression analysis usually are used to estimate non-combustion POC [Cabada et al., 2004]. The uncertainty in estimating SOC and POC usually arises from random measurement errors and the statistical techniques used to derive the primary OC to EC ratios.

Recently several groups evaluated linear regression techniques, such as linear least-squares, Deming regression, and York regression, which are often used in the EC tracer method to derive secondary and primary organic carbon [Chu, 2005; Saylor et al., 2006]. Chu [2005] concluded that Deming fit is better when the biomass burning contribution is high. Similarly, Saylor et al. [2006] found that when limited information is available on the relative uncertainties of OC and EC, then Deming regression is better. Our past experience indicates that the results by using Deming fit are similar to linear regression analysis when the mass loadings are high, which results in good linear correlations independent of the regression analysis methods. When the results by linear least-squares regression and Deming regression are very comparable, results by the linear least-squares analysis can be used. Most papers report results from linear least-squares. The caveat is that the linear correlation may fall apart when the particle mass loadings are low, especially approaching the instrument detection limits. This inevitably results in more scattered data and difficulty to derive more precise conclusions.

3.2 Other methods

Several methods are commonly used to derive SOC and POC, including the organic tracer-based receptor model [Schauer et al., 1996; Schauer et al., 2002], the reactive chemical transport model [Pandis et al., 1992; Strader et al., 1999], the non-reactive transport model [Hildemann et al., 1996] and the semi-empirical EC tracer method [Castro et al., 1999; Turpin and Huntzicker, 1995] detailed above. Yu et al. [2004] developed a hybid approach that combines the empirical primary OC:EC ratio method with a transport/emission model of OCpri and EC, to estimate the concentrations of SOC and POC, which is termed the emission/transport of primary OC:EC ratio method.

3.3 Comparison of SOC and POC

In this section, we will focus on a comparison between SOC and POC results from the AMS positive matrix factorization analysis (PMF) method and EC tracer method, both of which are being used widely. Results from newer measurement techniques, such as the Aerodyne Aerosol Mass Spectrometer (AMS) [Canagaratna et al., 2007] and the Particle-Into-Liquid Sampler coupled with Total Organic Carbon analyzer (PILS-TOC), were analyzed to derive secondary organic aerosols [Sullivan et al., 2006]. The approach used by Takegawa et al. [2006], to analyze the AMS data is conceptually similar to the semi-empirical EC tracer method; whereas secondary organic aerosol (SOA) formation was inferred from direct measurements of water-soluble organic carbon (WSOC) by PILS-TOC.

A two component PMF of the AMS data results in deconvoluted OOA (oxygenated organic aerosol), HOA (hydrocarbon-like organic aerosol [Lanz et al., 2007; Ulbrich et al., 2009]. Comparisons with other gas and aerosol phase measurements at an urban site in Mexico City during the MILAGRO campaign, namely T1, indicate that the HOA component reflects primary organic aerosols generated by combustion processes (i.e., vehicle emissions and some trash/biomass burning); while the OOA component reflects secondary organic aerosol species [de Gouw et al., 2009]. In order to make a meaningful comparison between the POC, SOC, and OC determined by the Sunset OCEC field analyzer and the AMS component mass concentrations, we calculate POA and SOA concentrations taking into account of the estimated OM/OC ratios of the two components, where OM refers to organic matter. Aiken et al. [2008] used the High Resolution ToF AMS measurements to obtain OM/OC ratios of 1.38, 1.95, and 1.55 for the HOA, OOA, and BBOA (biomass burning organic aerosol) components measured at the T0 site during the MILAGRO study. Since the HOA component at T1 is influenced by vehicle emissions as well as biomass burning, we estimate its OM/OC ratio to be 1.4, the average of the HOA and BBOA values determined at T0 (the other urban site closer to the downtown area in Mexico City); the OM/OC ratio for the T1 OOA component is estimated to be identical to the T0 value of 1.95.

Figure 3 depicts the comparison of AMS HOA, OOA, and OM vs. Sunset determined POA (POC*1.4), SOA (SOC*1.95), and OM (OM=POA+SOA), respectively. The Sunset POA, SOA, OM are in red, and the quantities determined by AMS in blue for HOA, OOA, and OM, respectively. Scatter plots of corresponding quantities by AMS and Sunset are also presented.

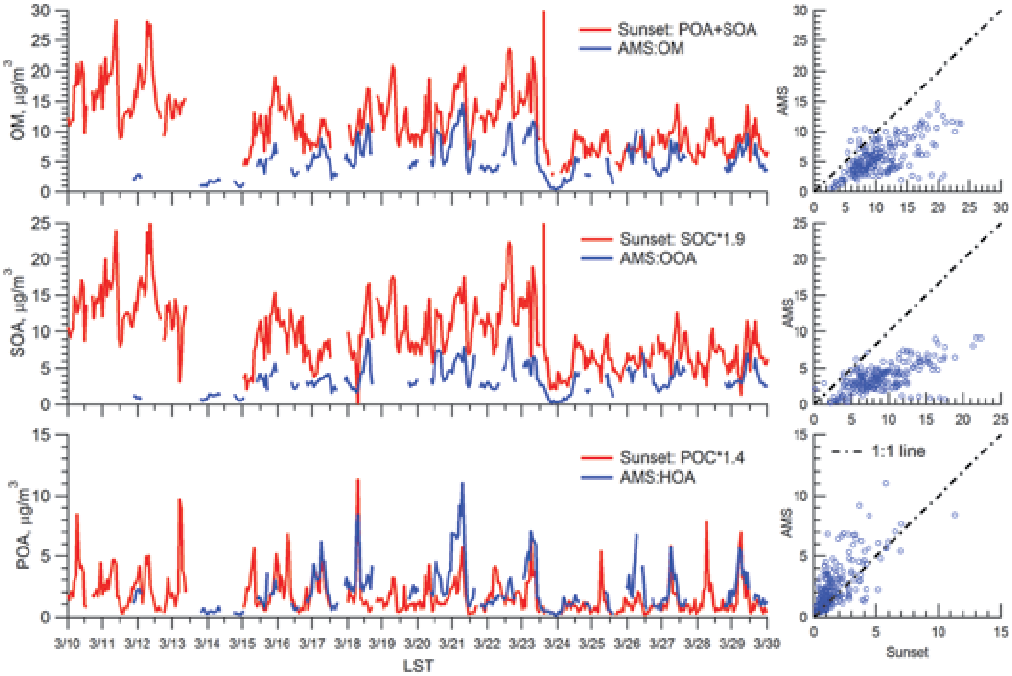

Fig. 3. Comparison of the AMS HOA, OOA, and OM vs. the Sunset POA, SOA, and OM at an urban site in Mexico City.

As to the OM comparison, several factors could contribute to these results. The first is the conversion factor used to convert OC to OM by the Sunset measurements. The Deming linear regression analysis of AMS total OM vs. Sunset OC results in a slope of 1.2±0.2. If 1.2 were used to convert the Sunset OC to OM, the difference of the total OM determined by the AMS and those by Sunset instruments is reduced. However, recent studies by the high resolution AMS indicate that the conversion factors for POA and SOA may not be the same [Aiken et al., 2008]. Therefore, we use the sum of POA and SOA to arrive at OM. Second the size cut of AMS and the Sunset OCEC differs. The former is approximately 1 μm and the latter 2.5 μm, which could contribute to the difference in total organic matter mass loadings.

As to POA, a comparison was made between the AMS HOA vs. POA (Sunset). The general trend between the HOA and POA is in agreement over the entire field study period. As to SOA, two sets of comparison were made: AMS OOA vs. SOA (SOA=SOC*1.95) and AMS OOA vs. SOA (SOA=SOC*1.4). One factor contributing to the difference is the conversion factor used to convert SOC to SOA. The factor determined by Aiken et al. [2008], i.e. 1.95, results in higher SOA compared with the factor 1.4 determined by an earlier review [Turpin et al., 2000]. Similarly, another factor contributing to the difference is size cut as discussed in the OM comparison. Since the OC emitted from non-combustion sources (vegetation etc.), as

well as emissions directly from biomass burning, are assumed to be negligible in the EC-tracer method, it cannot be used to derive BBOA. In future studies we should investigate the differences among different methods used to arrive at SOA and POA in more detail.
The Deming linear least-squares fit results in a slope of 0.8±0.1 for AMS OM vs. Sunset OM, 1.2±0.2 for AMS HOA vs. Sunset POA, 0.5±0.2 for AMS OOA vs. Sunset SOA (SOA=SOC*1.4), and 0.4±0.1 for AMS OOA vs. Sunset SOA (SOA=SOC*1.95).

4. Conclusion

Thermal desorption analysis method has been widely used for the determination of carbonaceous aerosols including TC, OC, and EC for decades. It is a proven technique. Compared to the newer single particle mass spectrometery or ensemble particle mass spectrometry, it is simple to operate. Data reduction is less complicated and labor intensive unlike the mass spectrometer data deconvolution, for example. It is useful for the community to compare different thermal optical protocols to clearly define the differences among them. This will undoubtedly improve the comparability among data sets utilizing different thermal optical methods.
It is equally useful to reach consensus about the measurement difference of EC using different techniques. More research has been conducted recently, it is time more conclusive solutions be reached to make data sets more useful for experimental intercomparisons and model input. For the purpose of waste management and monitoring, it is most needed to use inexpensive, easy to operate, fast on-line analytical methods. The established semi-continous Sunset OCEC field analyzer is a good option at present. However, a smaller, more portable version may make the application and measurement of carbon aerosols more accessible to the community. As we have shown more development has been made to the in situ measurement of EC or BC in the past decades. One success is the micro Aethalometer® (Magee, model AE51). The real challenge lies in the determination of OC. Newer techniques are needed to make this happen in addition to continued effort to improve existing ones.

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